ASSESSMENTS OF SEDIMENT QUALITY

During Phase I of the Elizabeth River Long-Term Monitoring Program, a number of biological, chemical, and toxicological investigations were conducted by investigators at Old Dominion University and the Virginia Institute of Marine Science. Results from these studies have been published in three previous technical reports (Dauer et al., 1989; Ewing et al., 1989; Greaves, 1990). This section summarizes data from a number of these studies through several approaches currently being employed in sediment quality assessments throughout the country: the Sediment Quality Triad; the Apparent Effects Threshold; and the Equilibrium Partitioning Approach. Each of the approaches provides visual summaries and "references" by which data sets can be compared.

In addition to the utilization of the three techniques in presenting the Phase I data, the results of more theoretical evaluations employing bootstrap simulations will be presented in a discussion of the advantages and limitations of each technique.

Sediment Quality Triad

Applications of Triad Approach to Elizabeth River Data

One method of displaying and summarizing sediment quality data is the sediment quality Triad (Chapman et al., 1987). Data from sediment assessment investigations are summarized as indices plotted on three axes: one representing sediment toxicity; a second representing _in situ_ biological effects; and a third representing sediment chemistry (contamination). The indices plotted on the axes represent "ratio-to-reference" (RTR) values, where the values for given parameters at a site are divided by the values of the same parameters measured at a reference site which is not contaminated. In this way, the relative degree of "impact" along each axis can be presented visually. Although this approach does have some theoretical drawbacks (see below), it does provide a rather effective graphical summary of the data from each site. Therefore, Triad plots will be presented for each of the 12 Elizabeth River site as a _preliminary_ evaluation tool.

In calculating the Triads, three sets of tables were prepared to document the relationship between the raw data and the RTR values. Table 23 presents the mean mortalities and RTRs from sediment (solid phase) bioassays of _Cyprinodon variegatus_ (fish), _Palaemonetes pugio_ (grass shrimp), and _Mytilus edulis_ (blue mussels). The RTR mean was the average of the RTR values from each test. As has been previously reported, little indications of sediment toxicity were observed with these particular species. It should be noted that more extensive assessments with more sensitive bioassay tests are currently being conducted in the Elizabeth River Management/Monitoring Phase II Study.

Table 24 presents the results of benthic macrofaunal studies. In order to make RTR values increase with increasing "impacts", the reciprocals of the various diversity and biomass measurements were used for RTR calculations (Chapman et al., 1987). The benthic

Table 23. Mean mortalities and RTR values for solid phase bioassays of *Cyprinodon variegatus*, *Palaemonetes pugio*, and *Mytilus edulis*. The reference values represent the means of three tests: one conducted on Thimble Shoal Channel sediments, and two on sediments from the Nansemond River.

	Fish		Shrimp		Mussels		
Site	Mean Mortality %	RTR	Mean Mortality %	RTR	Mean Mortality %	RTR	RTR Mean
ELI1	11	0.8	1	0.2	6	2.0	1.0
ELI2	6	0.4	5	0.8	0	0	0.4
ELI3	15	1.0	4	0.7	2	0.7	0.8
LAF1	7	0.5	6	1.0	2	0.7	0.7
WBE1	10	0.7	0	0	1	0.3	0.3
EBE1	10	0.7	2	0.3	0	0	0.3
EBE2	27	1.8	0	0	0	0	0.6
SBE1	4	0.3	1	0.2	0	0	0.2
SBE2	9	0.6	0	0	0	0	0.2
SBE3	17	1.2	1	0.2	3	1.0	0.8
SBE4	13	0.9	0	0	1	0.3	0.4
SBE5	8	0.5	5	0.8	5	1.7	1.0
REF	14.7	1.0	6	1.0	3	1.0	1.0

Table 24. Reciprocals of diversity and biomass measurements and RTR values for benthic macrofaunal analyses. The refernce data represent mean values observed over a four-year period at the Thimble Shoal Channel reference site.

Site	Richness (1/x)	Richness RTR	Diversity (1/x)	Diversity RTR	Evenness (1/x)	Evenness RTR	Biomass (1/x)	Biomass RTR	RTR Mean
ELI1	0.44	2.8	0.85	3.4	1.20	1.0	41.67	14.0	5.3
ELI2	0.47	2.9	0.60	2.4	1.16	0.9	100.00	33.6	10.0
ELI3	0.56	3.5	0.72	2.9	1.50	1.2	100.00	33.6	10.3
LAF1	0.00	0	0.00	0	0.00	0	1000	335.6	335.6*
WBE1	0.51	3.2	0.66	2.6	1.18	0.9	111.11	37.3	11.0
EBE1	0.57	3.6	0.61	2.4	1.09	0.9	111.11	37.3	11.0
EBE2	0.38	2.4	0.53	2.1	1.17	0.9	76.92	25.8	7.8
SBE1	0.45	2.8	0.79	3.2	2.09	1.7	7.46	2.5	2.6
SBE2	0.85	5.3	0.91	3.6	1.62	1.3	62.50	21.0	7.8
SBE3	0.64	4.0	0.79	3.2	1.27	1.0	142.86	47.9	14.0
SBE4	1.50	9.4	1.34	5.4	1.47	1.2	76.92	25.8	4
SBE5	0.69	4.3	0.96	3.8	1.06	0.9	15.38	1.0	1.0
REF	0.16	1.0	0.25	1.0	1.25	1.0	2.98	1.0	1.0

Note:

* Only 1 species and 2 individuals found, so no values could be calculated except biomass. Therefore, this value does not represent a true mean RTR.

sample collected at Site LAF1 had very few (two individuals of one species) macrofaunal organisms, so no diversity indices could be calculated.

Table 25 presents the results of the studies on organic contaminants (Greaves, 1990) and trace metals in sediments. Separate table sections were made for the metals (eight metals) and for the organic indices (TBT, aromatics, polar compounds, and PCBs). The RTR values plotted on the Triad graphs represent the grand mean of the mean metal and organic RTRs (Chapman et al., 1987).

The Triad graphs are presented in Figures 73-75. In each Triad plot, the three axes are scaled to accommodate the largest value plotted for any single RTR index in the entire 12-site series (extreme outliers are indicated but not included in the scaling to provide a more effective visual presentation). The small dotted triangle represents the unit-value reference site conditions, while the solid line triangle summarizes the RTRs for the particular site.

The sediment toxicity results for all sites were not significantly different from the controls, so the toxicity RTRs remain very close to unity. The lack of apparent toxicity in Phase I studies could be because the 12 sites did not coincide with some of the areas previously shown to be toxic (Alden and Young, 1982; Alden and Butt, 1988; Alden et al., 1988) and/or due to the fact that species selected as "representative indigenous fauna" were not as sensitive as test species now being used in Phase II.

Table 25 . Concentrations and RTR values for sediment contaminants: A) Metals (mg/kg) and B) Composite organic (ug/kg) categories.

A) Metals:

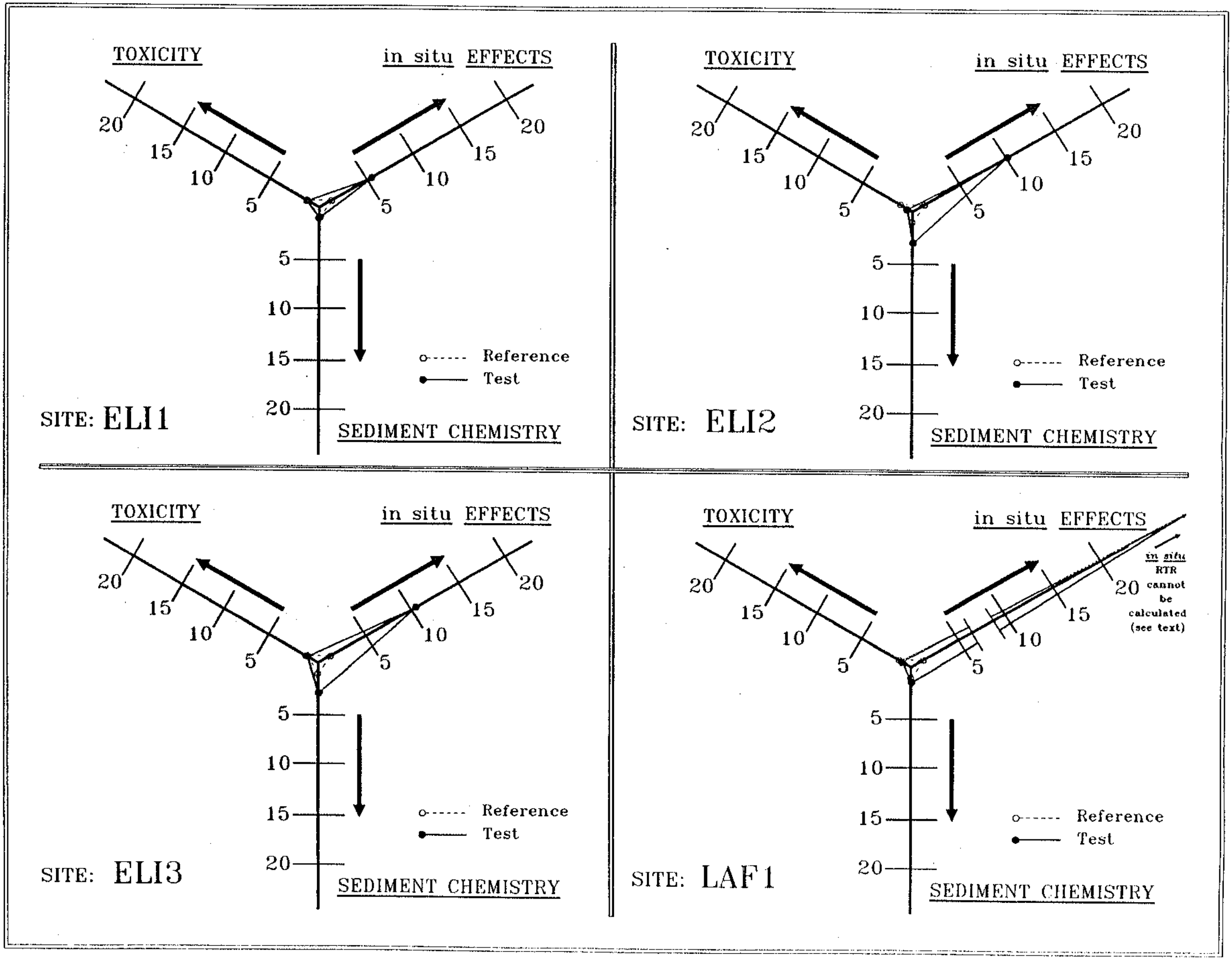
Site	Cd	Cd RTR	Cr	Cr RTR	Cu	Cu RTR	Pb	Pb RTR	Hg	Hg RTR	Ni	Ni RTR	Ag	Ag RTR	Zn	Zn RTR	RTR Mean
ELI1	0.00	0	43	1.3	22	1.2	34	1.0	0.13	0.8	19	1.2	0.6	1.2	116	1.1	1.0
ELI2	1.25	4.2	56	1.7	39	2.0	55	1.7	0.15	0.9	23	1.4	0.8	1.6	232	2.1	2.0
ELI3	1.79	6.0	32	1.0	39	2.0	82	2.5	0.20	1.2	13	0.8	1.1	2.2	267	2.4	2.3
LAF1	0.56	1.9	50	1.5	23	1.2	41	1.2	0.08	0.5	21	1.3	0.6	1.2	102	0.9	1.2
WBE1	6.31	21.0	54	1.6	70	3.7	129	3.9	0.34	2.0	18	1.1	1.1	2.2	666	6.1	5.2
EBE1	1.52	5.1	65	2.0	172	9.0	169	5.1	0.72	4.2	24	1.5	1.4	2.8	499	4.6	4.3
EBE2	0.77	2.6	38	1.2	150	7.9	300	9.1	1.25	7.4	17	1.0	1.2	2.4	467	4.3	4.5
SBE1	2.10	7.0	64	1.9	163	8.6	148	4.5	0.62	3.6	25	1.6	1.3	2.6	572	5.2	4.4
SBE2	1.75	5.8	68	2.1	186	9.8	159	4.8	0.75	4.4	36	2.2	1.3	2.6	495	4.5	4.5
SBE3	1.08	3.6	50	1.5	96	5.0	166	5.0	0.52	3.1	21	1.3	0.4	0.8	301	2.8	2.9
SBE4	0.55	1.8	37	1.1	55	2.9	65	2.0	0.25	1.5	15	0.9	0.3	0.6	192	1.8	1.6
SBE5	0.91	3.0	58	1.8	91	4.8	97	2.9	0.45	2.6	23	1.4	0.9	1.8	288	1.0	1.0
REF	0.30	1.0	33	1.0	19	1.0	33	1.0	0.17	1.0	16	16	0.5	1.0	109	1.0	1.0

B) Organics:

Site	TBT	TBT RTR	Aromatics	Aromatics RTR	Polar Fraction	Polar Fraction RTR	PCB's	PCB's RTR	RTR Mean	RTR Mean for Metals & Organics
ELI1	32	3.6	3100	2.1	<10	0.1	*	*	**	7.13
ELI2	99	11.0	5400	3.7	320	2.1	24	1.0	4.4	3.2
ELI3	39	4.3	6700	4.6	110	0.7	120	5.0	3.6	2.95
LAF1	15	1.7	1400	1.0	60	0.4	74	3.1	1.6	1.4
WBE1	19	2.1	6100	4.2	230	1.5	240	10.0	4.4	4.8
EBE1	1100	122.2	31000	21.3	1700	11.0	660	27.5	45.5	24.9
EBE2	220	24.4	60000	41.2	5500	35.5	400	16.7	29.4	16.85
SBE1	1533	170.3	19000	13.1	840	5.4	1213	50.5	59.8	32.1
SBE2	1830	203.3	47667	32.8	2330	15.0	853	35.5	71.6	38.0
SBE3	670	74.4	53000	36.4	3350	21.6	150	6.3	34.7	18.8
SBE4	318	35.3	45333	31.2	3267	21.1	187	7.8	23.8	12.7
SBE5	403	44.8	55000	37.8	3733	24.1	241	10.0	29.2	15.9
REF	9	1.0	1455	1.0	155	1.0	24	1.0	1.0	

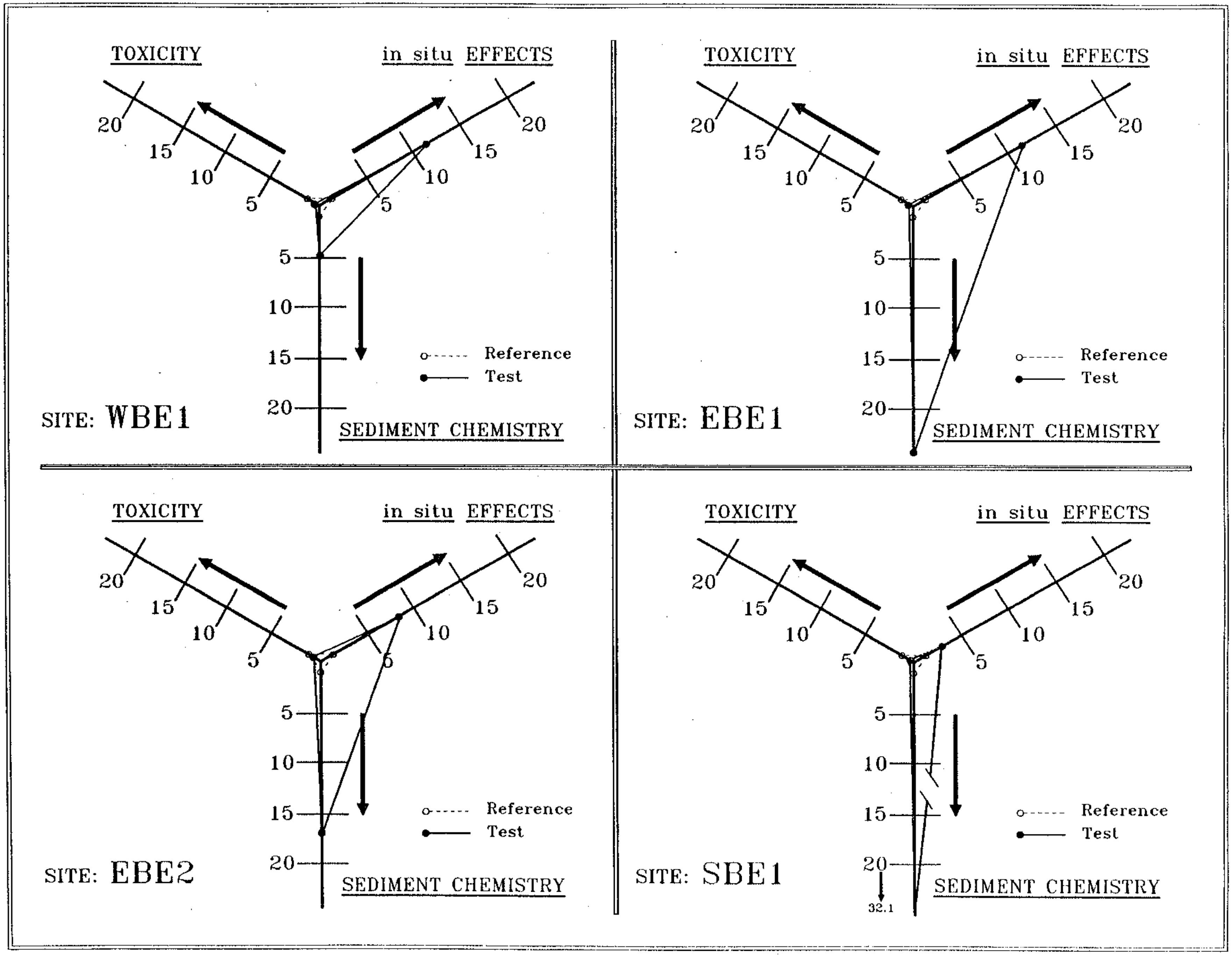
Notes: * = missing sample; ** = RTR mean does not include PCB RTR

Figure 73. Triad plots for Sites ELI1, ELI2, ELI3, and LAF1.



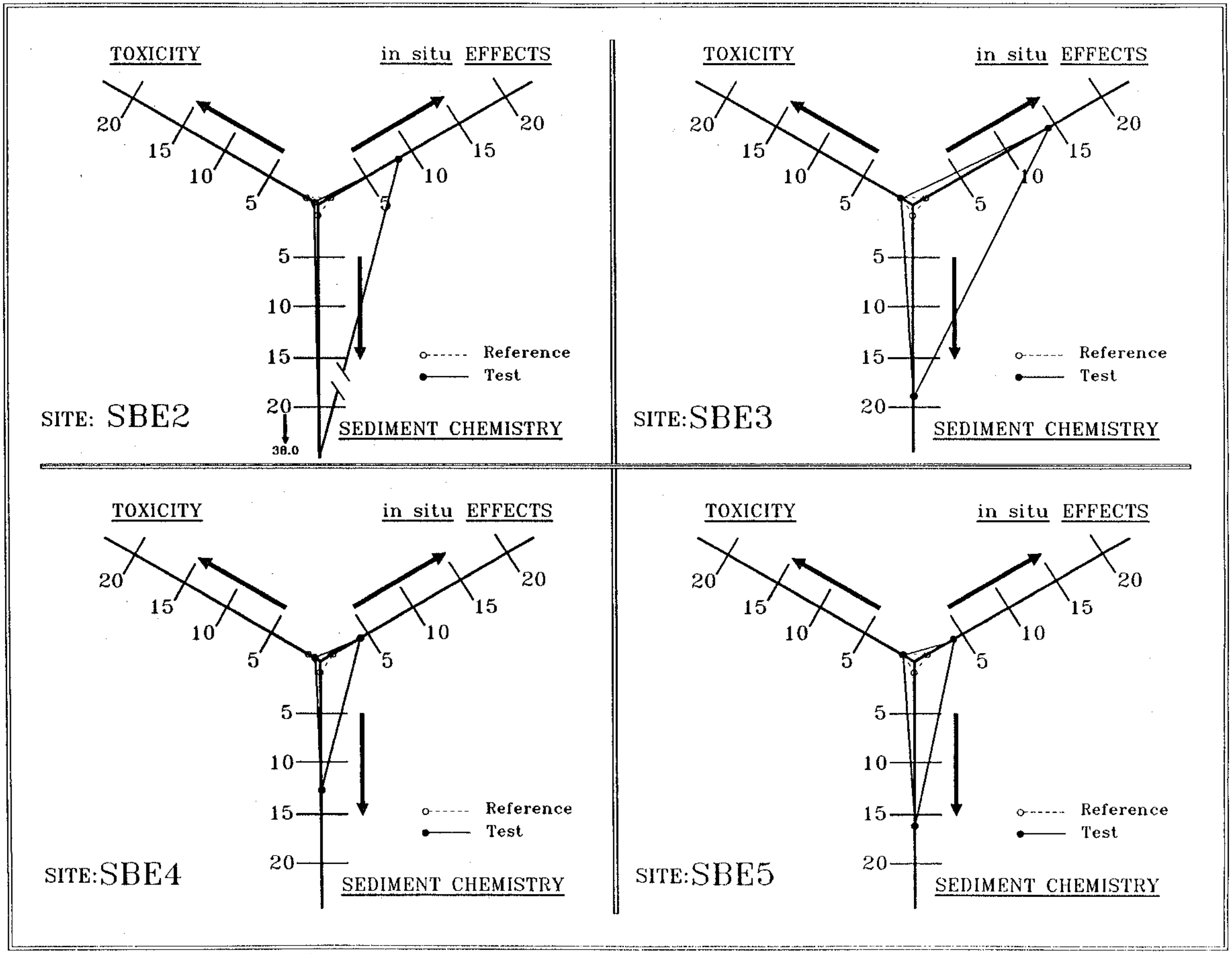
TOXICITY
in situ EFFECTS
SEDIMENT CHEMISTRY
Reference
Test
SITE: ELI1
TOXICITY
in situ EFFECTS
SEDIMENT CHEMISTRY
Reference
Test
SITE: ELI2
TOXICITY
in situ EFFECTS
SEDIMENT CHEMISTRY
Reference
Test
SITE: ELI3
TOXICITY
in situ EFFECTS
in situ RTR cannot be calculated (see text)
SEDIMENT CHEMISTRY
Reference
Test
SITE: LAF1

Figure 74. Triad plots for Sites WBE1, EBE1, EBE2, and SBE1.



TOXICITY
in situ EFFECTS
SEDIMENT CHEMISTRY
Reference
Test
SITE: WBE1
SITE: EBE1
SITE: EBE2
SITE: SBE1
32.1

Figure 75. Triad plots for Sites SBE2, SBE3, SBE4, and SBE5.


TOXICITY
in situ EFFECTS
SEDIMENT CHEMISTRY
Reference
Test
SITE: SBE2
38.0
SITE:SBE3
SITE:SBE4
SITE:SBE5

The RTRs for the _in situ_ and sediment chemistry axes were considerably higher than unity. It should be noted that the _in situ_ data from the Thimble Shoal Channel site may not be the most representative of all potential reference sites for the Elizabeth River sites because it is somewhat coarser in sediment texture and higher (possibly more stable) in salinities than many of the Elizabeth River sites. Therefore, until a reference site that is more closely "matched" to the physical/geological characteristics of the Elizabeth River can be found (and demonstrated to be uncontaminated) the RTRs along the _in situ_ axis should be considered for relative inter-site comparisons, rather than as an absolute deviation from unity. Sites in the Nansemond River appear to be promising alternatives for future references, but sediment chemistry data were not available from these areas until after Phase I studies were completed (the Thimble Shoal Channel site had been demonstrated to be "uncontaminated" in numerous previous studies).

In examining the Triad plots, it appears that the mainstem (ELI1-ELI3) sites were less "impacted" than the Eastern or Southern Branch sites. The Lafayette River site was unusual in that the benthic sample contained very few macroinfaunal organisms. However, due to extremely shallow depths, the research vessel deploying the box corer could not sample this site, so hand corer samples were collected by scuba divers. Although the approximate area and depth sampled by the hand corers were approximately the same as those observed for the box corer, the two methods may not have been comparable in collection efficiency. Therefore, the low diversity observed at LAF1 may have been an artifact of the sampling method rather than an index of the <u>in situ</u> "impact". The low degree of sediment contamination and absence of toxicity tend to confirm this speculation.

The sites of the Eastern and Southern Branches of the Elizabeth River all displayed greater signs of "impact" particularly along the sediment chemistry axis. Sites SBE1 and SBE2 displayed the greatest RTR sediment chemistry values, primarily due to organic contamination (Table 25). However, Sites EBE1 and SBE3 displayed high RTRs along both the <u>in situ</u> and the sediment chemistry axes, and therefore may be considered to be the most "impacted". Dauer et al. (1989) have shown that benthic populations in the Southern Branch appear to be depressed in diversity and biomass, while numerous studies have shown this region to be contaminated in metals (Johnson and Villa, 1976; Alden et al., 1981, Rule, 1986) as well as organics (Alden and Hall, 1984; Bieri et al., 1986; Alden and Butt, 1988).

<u>Assessment of Triad Approach</u>

One of the theoretical drawbacks to the Triad method of data presentation is that it does not display any indication of the degree of confidence that the researcher would place on the RTR indices. In fact, the very process of making a ratio of data from two different populations produces non-normal and possibly intractable distributions, so the usual methods for calculating confidence limits become impossible. Green (1979) provides a detailed discussion of the statistical consequences and possible spurious interpretations which may result from the analysis of data that have been ratioed.

A second possible drawback to the Triad approach to data presentation is one that is common to all simplifying/summarizing methods of visual display of results: a Triad plot for a single sample (and reference) looks identical to one summarizing an extensive long-term study of a region. Readers of primary source articles are usually well aware of the degree of confidence that they would place in data sets from any given study (particularly if statistical comparisons are presented for each of the components of the Triad). On the other hand, in the processes of review and assessment by environmental managers, the Triad plots may become "decoupled" from the source data or from any caveats concerning the nature of the data sets (e.g. such as those indicated above concerning the <u>preliminary</u> nature of the Triad plots produced for the Elizabeth River). Triad plots produced by contractors for single samples in a short-term study should not be viewed with the same degree of "confidence" as those summarizing a more comprehensive study with statistically validated conclusions. However, there is a potential temptation for environmental contractors to avoid the time and expense of collecting replicated data in a monitoring/assessment program with a statistically defined experimental design if the environmental manager will make decisions based upon a simple plot. If the Triad plot could display some indicator of the degree of statistical confidence that the data set actually merits, this temptation would be reduced, as would the potential for ill-advised management decisions. Thus, the limitation of the presentation mode in not allowing the display of confidence limits also increases the potential misuse/ misinterpretation of the technique.

A simulation study was designed to address the issues of single versus multiple observations, and the problem of the data distributions produced by ratios, as well as to demonstrate a technique for placing confidence limits on the Triad plots. For demonstration purposes, only the "in situ effect" portion of the Triad was considered. Data sets for the simulated Triad assessments came from two sources: benthic biological data from a baseline survey of Hampton Roads Harbor which was conducted from 1979-1983 by Hampton Roads Sanitation District (HRSD, 1984; Alden and LeBlanc, 1990) was used as the "reference" data set; and information from a 1979 study conducted in the Southern Branch of the Elizabeth River by Hawthorne and Dauer (1983) was used for the "impacted" data set. Since only summary data (means and standard deviation of benthic biological diversity and abundance indices) were available from the latter study, a simulation program (Alden, 1984) was employed to create a data set with statistical parameters (means, standard deviations) of the Elizabeth River data, but with the overall frequency distributions of the variables from the HRSD data set.

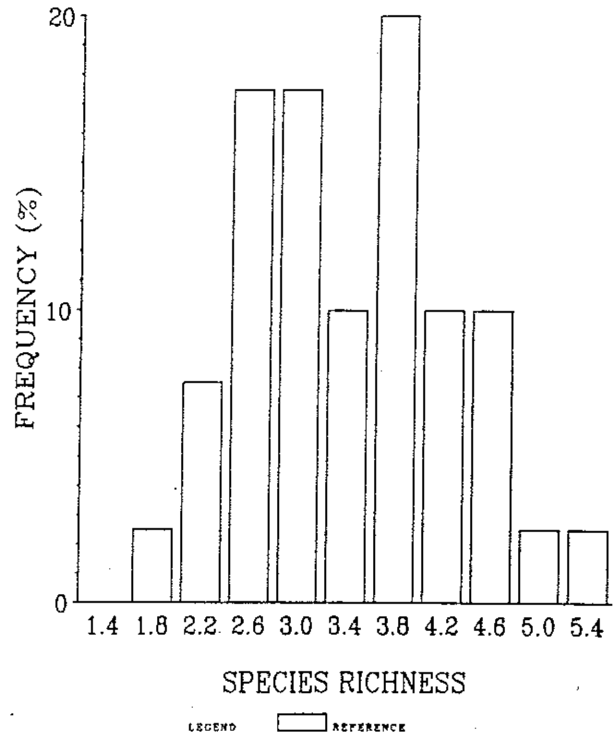
Boesch (1973) discusses the adverse impacts of multiple pollution sources on the benthic communities of the Elizabeth River compared to those of Hampton Roads Harbor during 1969. Adverse effects in the overall health of the benthic communities of the Southern Branch are apparent over 20 years later (Dauer et al., 1989), so the "impacted" data set representing this region would appear to provide a realistic example for the Triad assessment.

The species richness (Margalef's index) and the abundance (number individuals/m^2) were selected as the variables to be evaluated for those two data sets. These are two of the most commonly used indices of benthic community health. In addition, they present two very different frequency distributions: the species richness indices were normally distributed (Fig. 76), while the abundance data were highly skewed in distribution (Fig. 77). Analysis of variance demonstrated that differences between the "impacted" and "reference" data sets were very highly significant ($p<0.001$) for both of these variables (abundances were log-transformed prior to testing). The mean species richness was 2.4 (s.d.= 0.5) for the impacted and 3.4 (s.d.= 0.8) for the "reference" area. The mean abundances were 866 (s.d.= 541) and 3,888 (s.d.= 3.519) for the "impacted" and "reference" areas, respectively. Therefore, the assumption would be that an investigator using the Triad, or any variation on this approach, should detect a difference between the two areas in terms of <u>in situ</u> effects.

The first simulation involved the repeated selection of individual samples from both the "impacted" and "reference" data

Figure 76. Frequency distributions of Margalef's species richness measurements of benthic collections from reference (Hampton Roads Harbor) and impacted (Elizabeth River Southern Branch) areas (n=40).

FREQUENCY DISTRIBUTION:
SPECIES RICHNESS FOR REFERENCE AREA



FREQUENCY DISTRIBUTION:
SPECIES RICHNESS FOR IMPACTED AREA

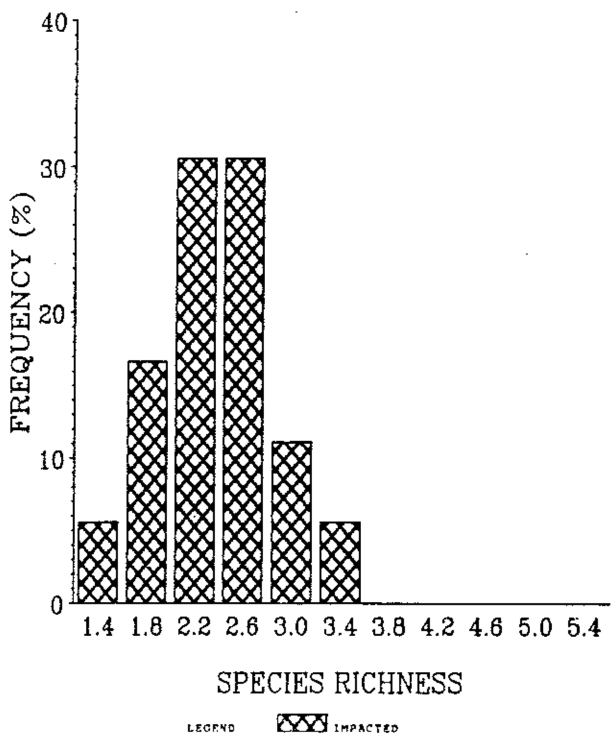
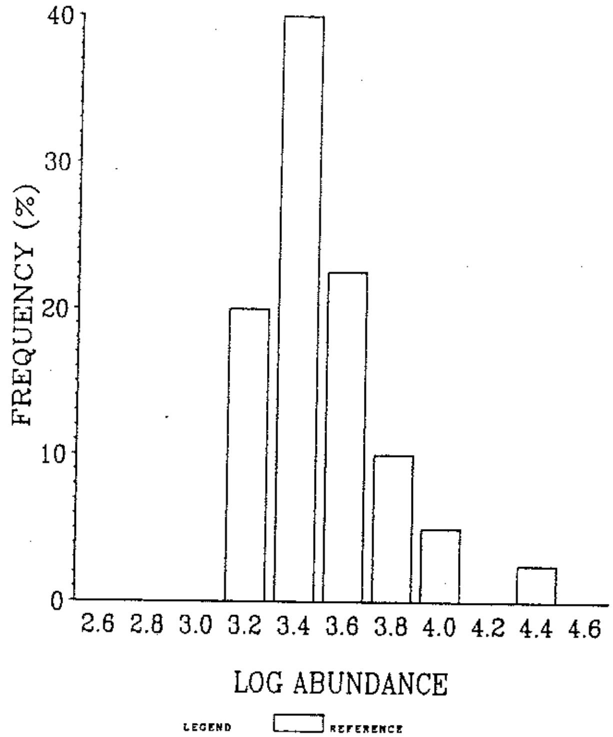
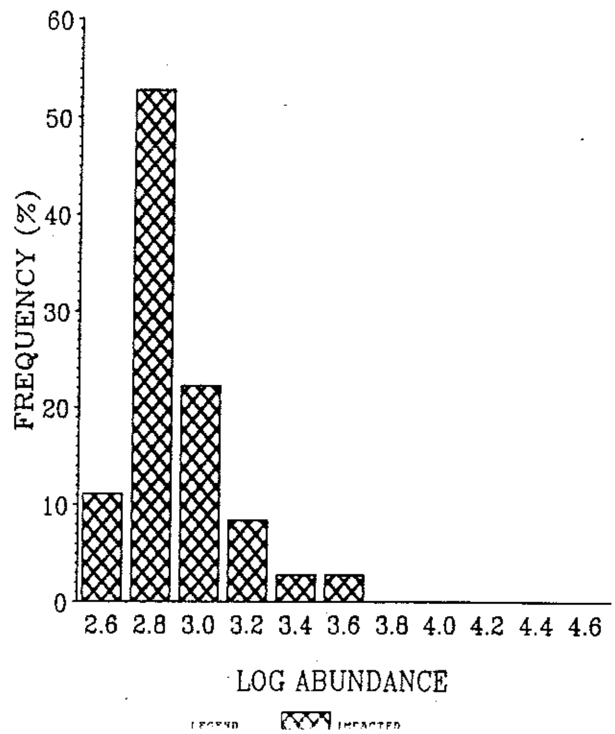


Figure 77. Frequency distributions of log abundance measurements (log no. individuals/m^2) from benthic collections from reference (Hampton Roads Harbor) and impacted (Elizabeth River Southern Branch) areas (n=40).

FREQUENCY DISTRIBUTION:
LOG ABUNDANCE (Ind/m.sq.) FOR REFERENCE AREA



FREQUENCY DISTRIBUTION:
LOG ABUNDANCE (Ind/m.sq.) FOR IMPACTED AREA

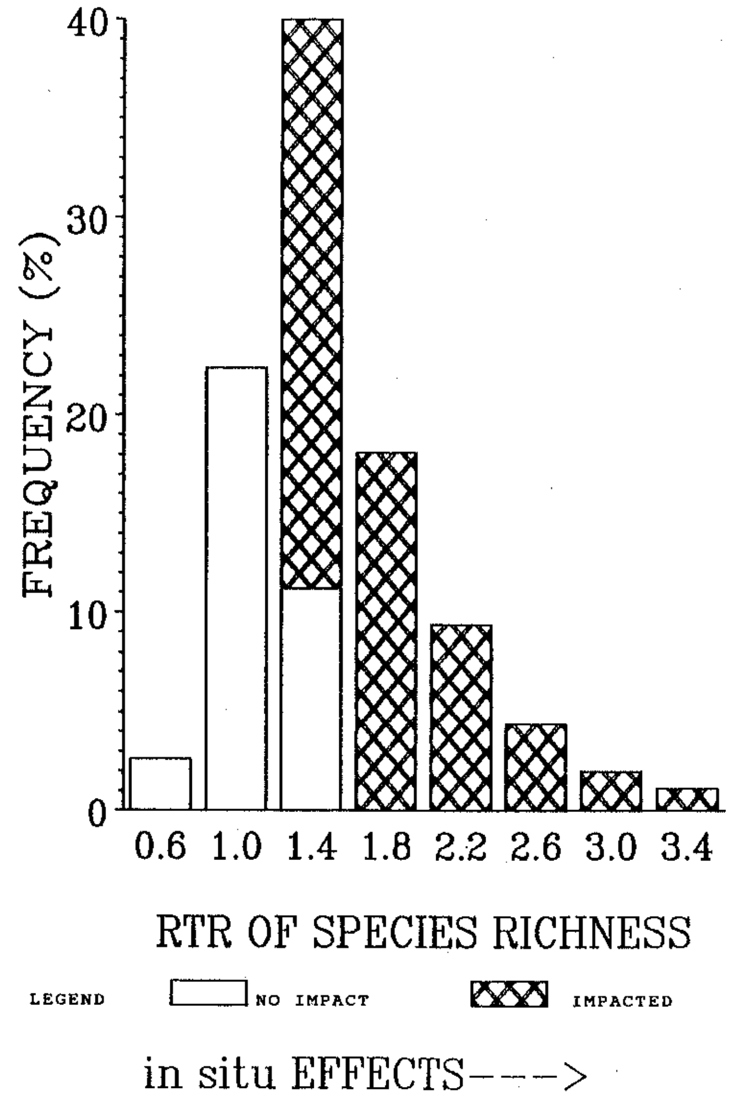


sets. Although both studies were conducted during the same relative period of time (overlap in 1979), they were conducted by independent research teams in the two areas, so there is no convenient way to link individual samples together. Therefore, a random selection process was adopted. The simulated pairs of "samples" were to represent a "one time" look at benthic communities from both areas, and RTR values were calculated. The RTRs were calculated as the reciprocals of species richness or abundance from the "impacted" sample divided by those of the "reference" area sample in order to produce scales that increase with increasing degree of impact (Chapman et al., 1987). The process was repeated 1,000 times to provide an estimate of the probability of "reaching the right conclusion" concerning the character of the areas based upon single samples.

Figure 78 presents the frequency distribution of the RTRs for species richness. Although the distributions of species richness measurements from both areas did not significantly deviate from normality, the RTR distribution appears to be skewed. Furthermore, if one assumes that a RTR would have to be greater than 1.2 (indicated by shaded portion of bars in Fig. 78) to declare an area different from the reference (RTR=1), there would be a 36% probability of concluding "no impact". It should be noted that this estimate of "Type II" error is likely to be conservative (low-end), because, if the samples were selected pair-wise by season, differences would likely have been less (i.e. some degree of common

Figure 78. Frequency distribution of ratio-to-reference (RTR) values for species richness. The open bar segments represent those values falling below an RTR value of 1.2.

FREQUENCY DISTRIBUTION:
SPECIES RICHNESS – RTR STANDARDIZED



seasonality would have been taken into account) and the resulting RTRs would have been smaller.

The results of the simulations of the abundance variable indicated a highly skewed RTR distribution (Fig. 79). However, the probability of concluding "no impact" was only 4.5%, probably due to the marked difference of the benthic abundance data from the two areas.

These results infer that it is dangerous to draw conclusions from a Triad presentation of single variables from single samples. Although RTRs based upon variables that are extremely different between the two areas may be correctly interpreted, those that are based upon other variables for which the differences are somewhat less dramatic, but still very highly significant, may lead to erroneous conclusions. When RTR values from a number of different variables are averaged, as is typically done (Chapman et al., 1987), the ultimate conclusions are less predictable. In the case of the simulation study, the composite <u>in situ</u> benthic health RTR which was calculated as the average of species richness and abundance RTRs was dominated by the magnitude of the latter values (Fig. 80). However, it could be speculated that a number of variables which behave like the species richness case (i.e. variables that are highly significantly different between two areas <u>statistically</u> but which involve a high Type II error when ratioed from single samples as RTRs) could produce composite RTRs that also lead to the "wrong" conclusion. Since these variables are not independent of each other, the probabilities cannot be calculated

Figure 79. Frequency distribution of ratio-to-reference (RTR) values for benthic faunal abundance. The open bar segments represent those values falling below an RTR value of 1.2.

FREQUENCY DISTRIBUTION:
BENTHIC ABUNDANCE – RTR STANDARDIZED

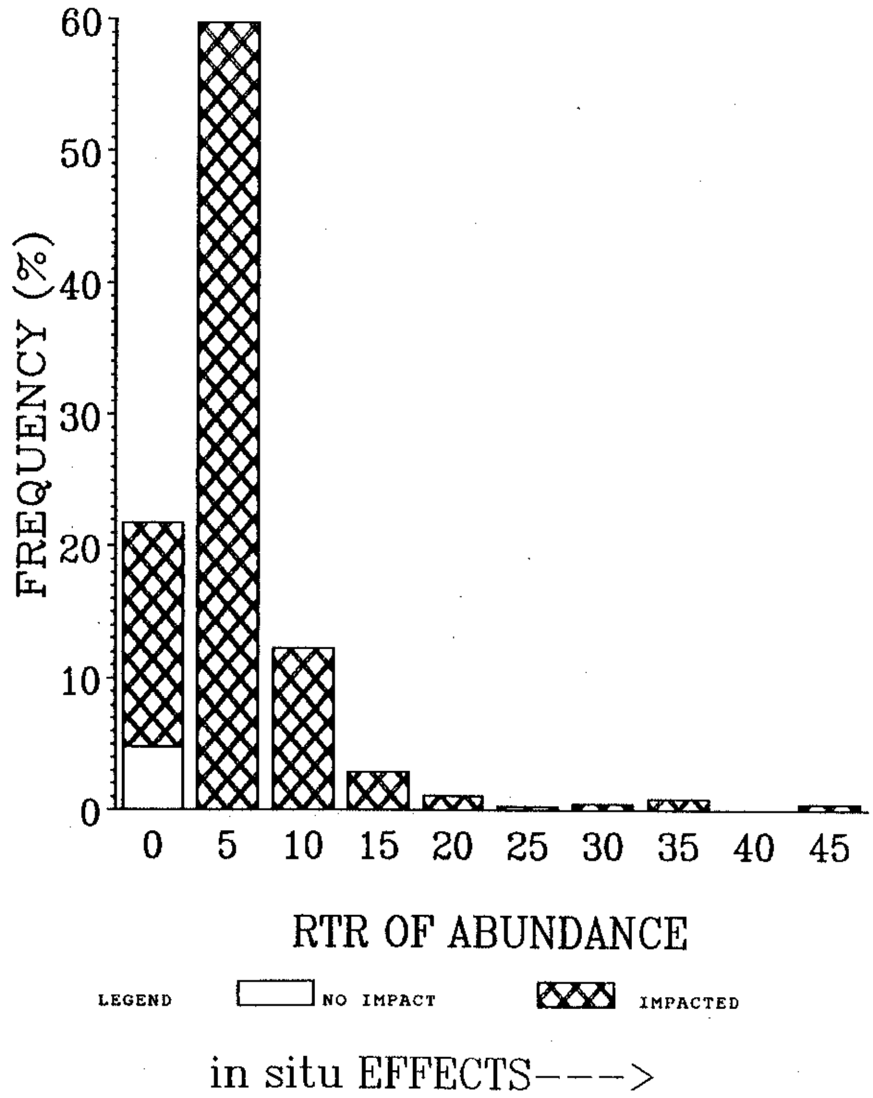
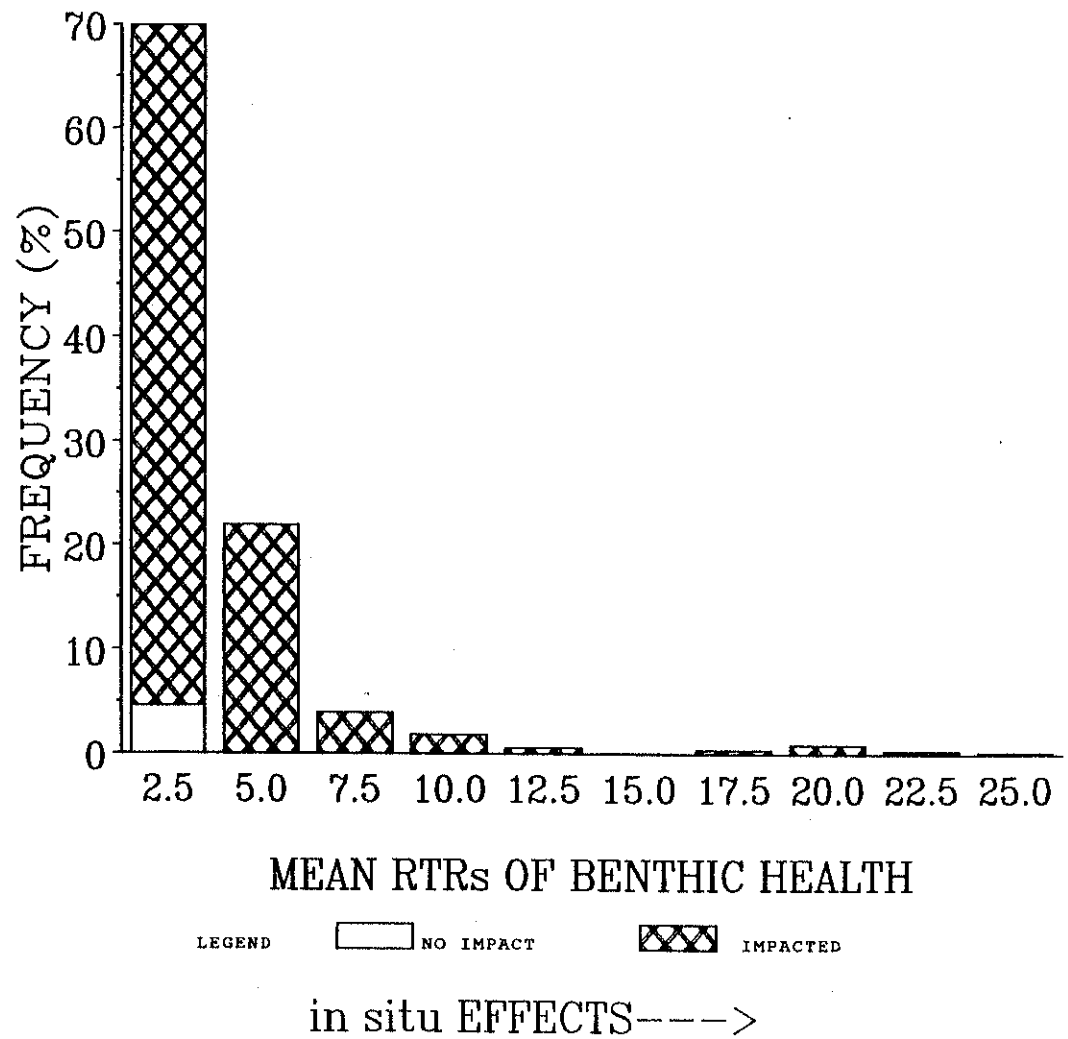


Figure 80. Frequency distribution of mean ratio-to-reference (RTR) values for species richness and abundance. The open bar segments represent those values falling below an RTR value of 1.2.

FREQUENCY DISTRIBUTION:
MEAN RTRs OF BENTHIC HEALTH
MEAN RTRs OF SPECIES RICHNESS & ABUNDANCE



directly, but could be estimated for individual conditions by simulations. Regardless of the situation, the degree of confidence that one can place on Triad plots representing a few samples is questionable. Furthermore, the non-normal distributions of RTRs make it difficult, if not impossible, to calculate classic confidence limits on RTR values which summarize even the most in-depth studies.

The second simulation study focused upon demonstration of a method to provide confidence limits for Triad plots. The approach involved a three step process. In the first step, variables (actually the reciprocals of the variables, see above) from the combined set of "impacted" and "reference" data were unit-deviate standardized (i.e. to produce standardized variables with a grand mean of zero and a standard deviation of one). As a result of this process, the "reference" values tended to be negative and the "impacted" values tended to be positive. The second step involved "bootstrap" simulations to produce confidence limits for the "impacted" and "reference" groupings within the data set (see Efron, 1979a,b; and Diaconis and Efron, 1983 for a detailed discussion of bootstrap techniques). The computer assembled "new" data sets containing 40 randomly selected samples (the size of the original data sets) from both the "impacted" and "reference" groups and calculated mean values for each. The simulation process of creating new data sets was repeated 1,000 times. The resulting output represented an approximation of the "universe" of possible mean values that could be produced by the data distributions of the two groups. Thus, the grand mean and probabilities representing confidence limits (e.g. 95%, 99%, etc.) could be empirically determined (e.g. the values between which 95% of all means fell, etc.) without assumptions concerning the data distributions. Although the unit deviate standardization does tend to "normalize" the data, normality is not necessary for the bootstrap approximations of confidence limits to work. There are considerations which may suggest that standardization to a common standard deviation may not be the most desirable approach (see below), but the technique can also provide confidence limit estimates without having the data unit deviate standardized.

The final step in the process involved scaling the data to the Triad axes. To produce a composite variable which may be summarized on a single axis, the values of the simulated means of the two variables were averaged together as the simulation process progressed. The means and confidence limits for the variables (or composite variable) were scaled so that the grand mean of the "reference" data was one. Since the means of the variables for "reference" areas were negative, this scaling was accomplished by adding the absolute value of the "reference" grand mean plus one to each of the means and confidence limits from both data sets. The distribution of the means for the scaled, standardized species richness, abundance and composite benthic community health indices are presented in Figures 81-83, respectively. These figures illustrate that there is no statistical overlap between the two areas for any of these variables. The obvious indication of

Figure 81. Frequency distributions of the means from 1000 bootstrap simulations of the reference (open bars) and impacted (cross-hatched bars) data sets for benthic species richness.

FREQUENCY DISTRIBUTION OF MEANS:
STANDARDIZED SPECIES RICHNESS
FROM 1000 BOOTSTRAP SIMULATIONS

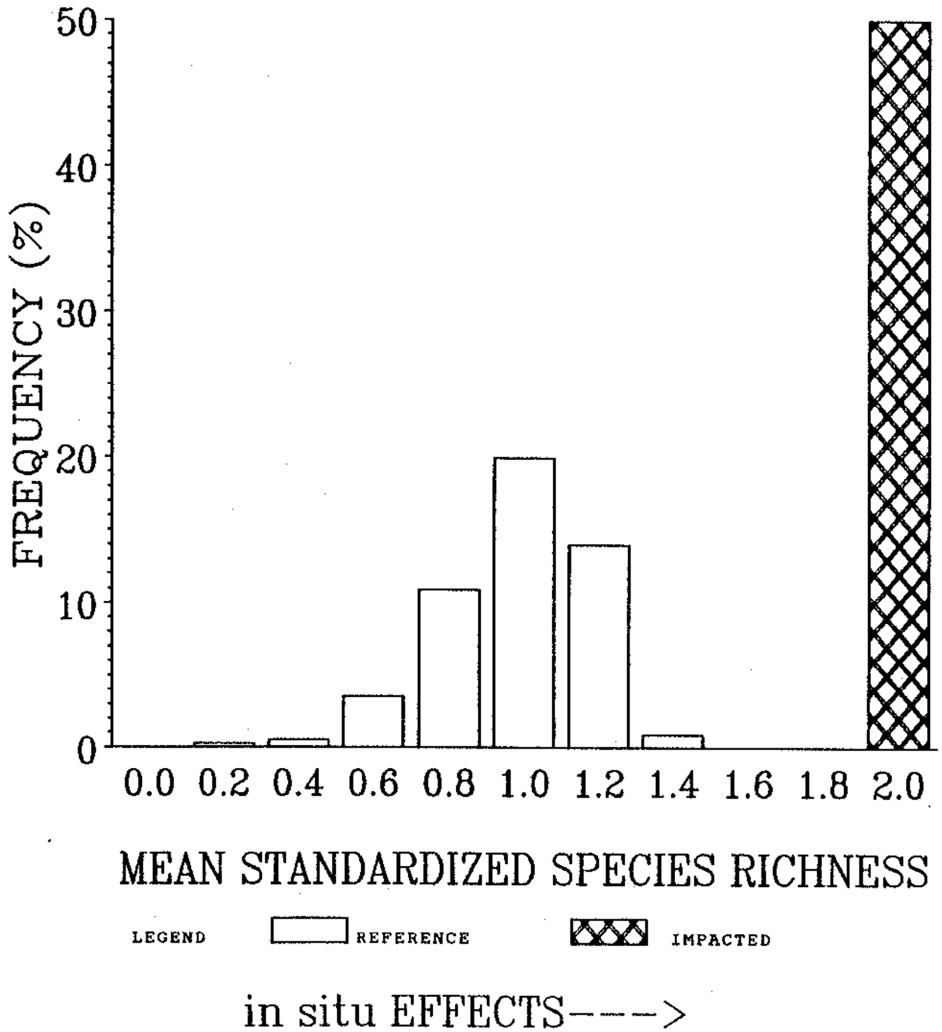
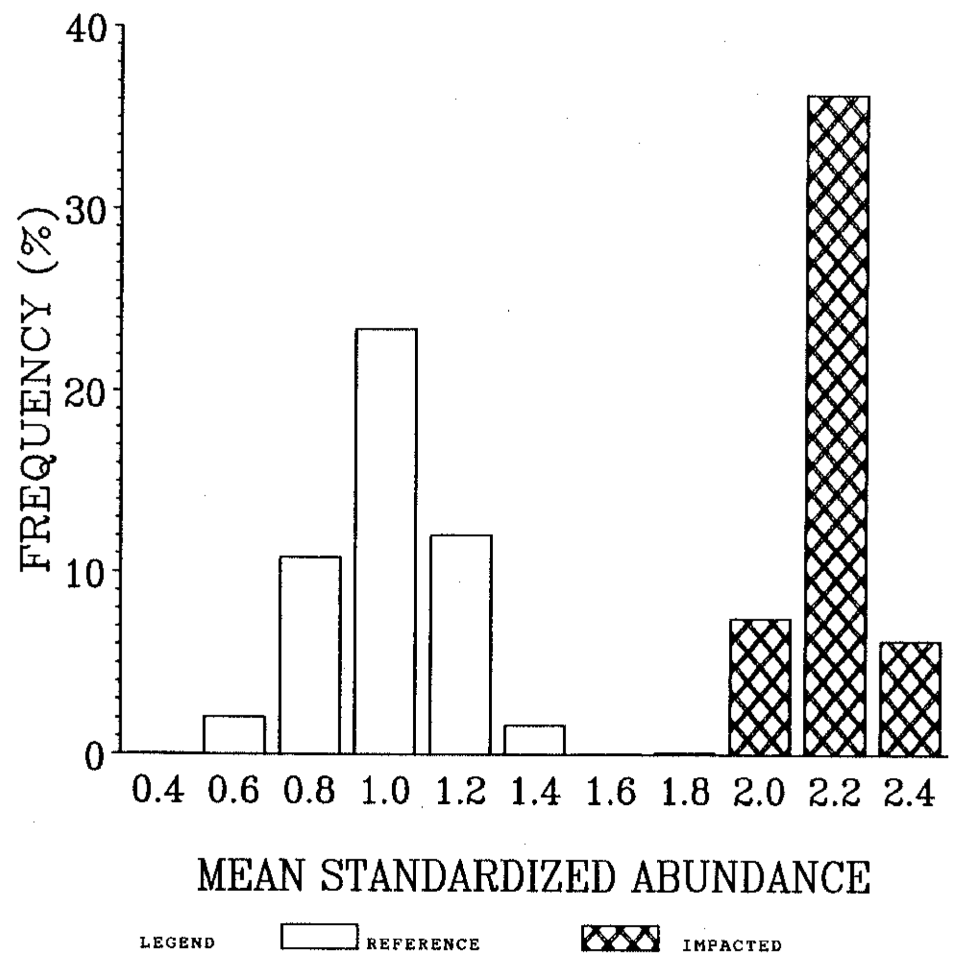


Figure 82. Frequency distributions of the means from 1000 bootstrap simulations of the reference (open bars) and impacted (cross-hatched bars) data sets for benthic biological abundance.

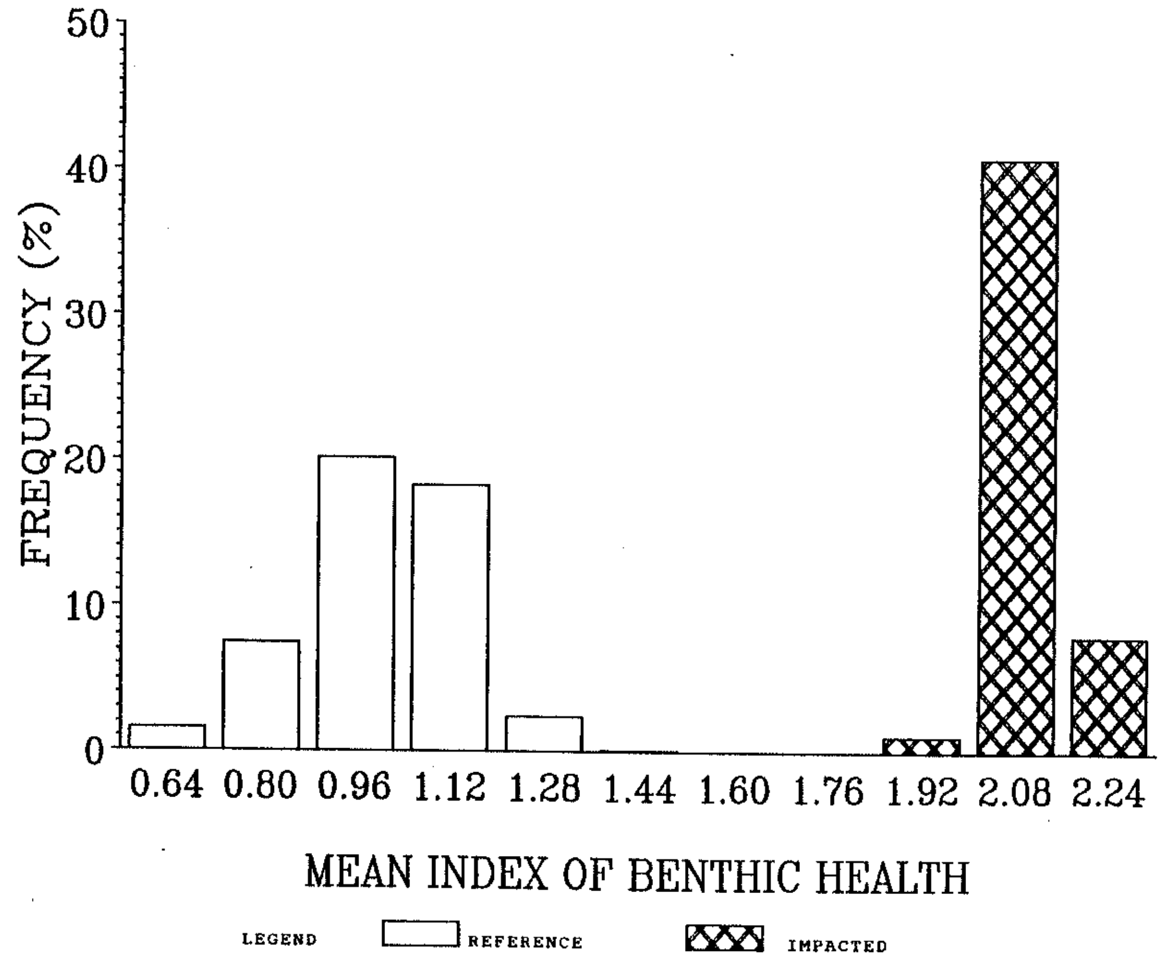
FREQUENCY DISTRIBUTION OF MEANS:
STANDARDIZED ABUNDANCE
FROM 1000 BOOTSTRAP SIMULATIONS



in situ EFFECTS--->>

Figure 83. Frequency distributions of the means from 1000 bootstrap simulations of the reference (open bars) and impacted (cross-hatched bars) data sets for a composite variable representing the mean of species richness and benthic abundance variables.

FREQUENCY DISTRIBUTION OF MEANS:
STANDARDIZED INDEX OF RICHNESS & ABUNDANCE
FROM 1000 BOOTSTRAP SIMULATIONS



in situ EFFECTS--->

statistical separation is also reflected in Table 26 which presents the mean and 95% confidence limits for each of the indices.

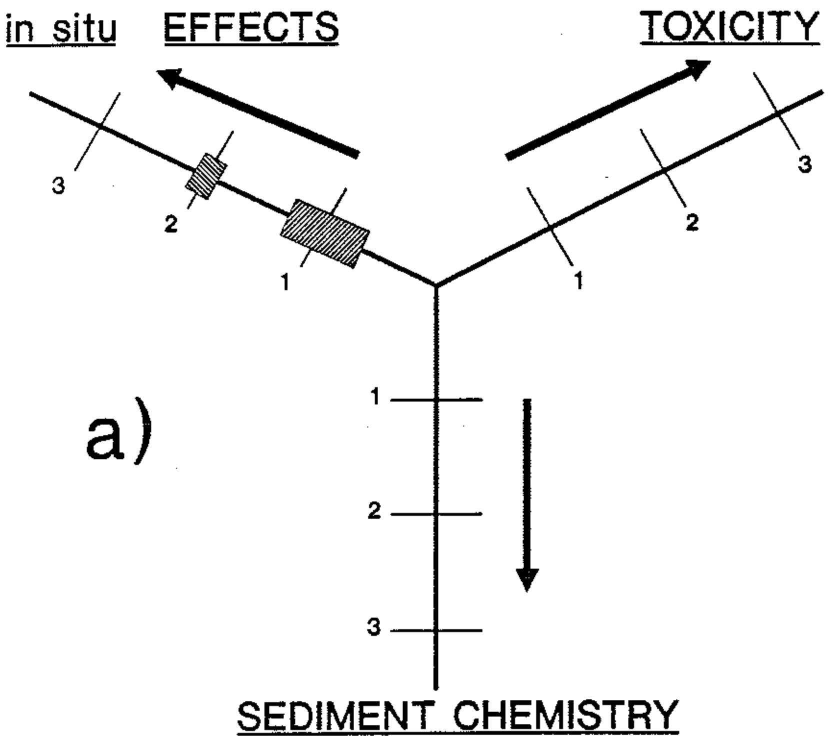
Table 26. Means, lower 95% confidence limits (LCL) and upper 95% confidence limits (UCL) determined by bootstrap simulations of scaled, standardized indices of benthic community health for "reference" and "impacted" areas.

	Reference Area			Impacted Area		
Variable	Mean	LCL	UCL	Mean	LCL	UCL
Species richness	1.00	0.64	1.26	2.00	1.95	2.05
Abundance	1.00	0.73	1.27	2.20	2.05	2.34
Composite Index	1.00	0.77	1.20	2.11	2.02	2.19

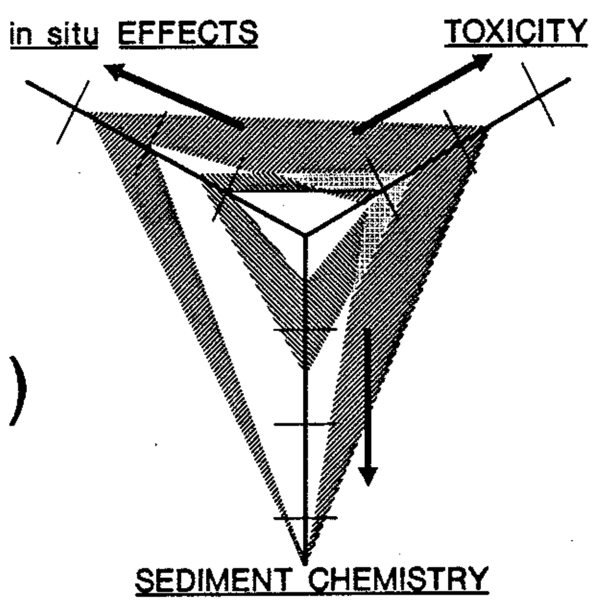
Figure 84a indicates the location of the means and confidence limits for species richness on a single axis of the Triad. Figure 84b presents a hypothetical Triad displaying confidence limits in terms of zones which quickly indicate which of the axes should be interpreted with the greatest degree of certainty.

One point should be made concerning the use of unit deviate standardization. The division of data from both "impacted" and "reference" data sets by the grand standard deviation of both data sets combined produces a new index which is standardized in magnitude. This standardization method has a couple of inherent

Figure 84. a) Display of 95% confidence limits for species richness (see Table 25) on the *in situ* axis of a Triad plot; b) hypothetical Triad plot displaying confidence zones on all three axes.


in situ EFFECTS
TOXICITY
SEDIMENT CHEMISTRY
1
2
3
1
2
3
1
2
3

a)


in situ EFFECTS
TOXICITY
SEDIMENT CHEMISTRY

b)

advantages: composite indices plotted on the axes are created from variables which are given relatively equal weight; and each axis tends to be "scaled" to the same order of magnitude as the others. Of course, these characteristics may also be seen as disadvantages, since the researcher may wish a variable (or entire axis) that is demonstrating a great level of difference between the test and reference areas to dominate the analysis. This sort of debate as to whether the "big picture" is best presented by the equal input of all conditions or by the influence of the dominating variable arises in a variety of disciplines when the subject of standardization of data is discussed. Fortunately, the approach (Steps 2 and 3) outlined above for producing confidence limits does not depend upon unit deviate standardization, so the researcher can select the avenue of choice for each situation. However, compositing of multiple variables to form indices becomes problematic if some sort of scaling (or transformation) is not considered. Of course, the bootstrap approach to the development of confidence limits can be applied to the traditional RTR calculations since it is not dependent upon any theoretical distributions (t, F, chi-square, etc.), and is, thus, immune to many of the statistical problems posed by such ratios. Therefore, even Triad plots based upon traditional RTRs can be displayed with confidence limits.

Apparent Effects Threshold

Application of AET Approach to Elizabeth River Data

Another method for the assessment of sediment quality involves the calculation of Apparent Effects Threshold (AET) values for the major contaminants. Tetra Tech (1986) details the methodologies for calculation of the level of a contaminant above which adverse effects (e.g. toxicity in sediment bioassays with a given species) always occur. There are some major theoretical limitations to the interpretations and/or applications of AETs, particularly when contaminants in the sediments are correlated with each other (see below). However, AETs may prove useful in identifying regions/ contaminants that merit further (confirmation) studies.

Due to the lack of significant mortalities in the sediment bioassays, AET values could not be calculated for the 1989 data base. However, AETs were calculated for metals collected in 1979 as part of a U.S. Army Corps of Engineers study (Alden et al., 1981; Alden and Young, 1982; Rule, 1986). These AETs are presented in Table 27, along with the sites sampled in the 1989 Phase I study and the 1989 sediment contaminant distribution study (surface samples from 0-10 cm depth) that had sediments with concentrations exceeding these "AETs". The sites from the former study exceeding the 1979 AETs for metals were generally those identified in the Triad presentation as being the most "impacted": Sites EBE1, EBE2, SBE1, SBE2, and SBE3. The only exception was WBE1, which exceeded the AET for cadmium and also displayed elevated levels of zinc and

Table 27. Apparent Effects Threshold for Metals in Sediments.

Metal	AET (mg/kg)	% Above AET and (Maximum Concentration) 1979	1989	Sites* in 1989 Exceeding 1979 AETs
Cd	3.7	1.8 (7.0)	2.0 (6.3)	WBE1
Co	-	-	-	-
Cr	68	3.8 (78)	5.9 (76)	SBE1, SBE2
Cu	142	5.1 (478)	29.4 (987)	SBE1, SBE2, EBE1, EBE2, EB, K (RS), L, M
Fe	46,043	0.1 (61,660)	0 (40,220)	-
Pb	176	5.6 (477)	17.6 (600)	SBE3, EBE2, EB,L,M (C,RS)
Mn	-	-	-	-
Ni	-	-	-	-
Zn	860	1.9 (1,150)	0 (666)	-

* surface sediments (0-10 cm) only.

lead. However, due to the relatively low level of organics contamination at WBE1, sediment chemistry RTR was not extremely high in the Triad presentation. The sites from the sediment contaminant distribution study that exceeded the AETs were those considered to be "hot spots" by spatiotemporal evaluations in the sections entitled **Metal Contaminants** and **Long-Term Temporal Patterns:** Sites EB, K, L and M.

Assessment of the AET Approach

There are a number of limitations to the AET approach which have been discussed elsewhere (e.g. Tetra Tech, 1986). To summarize, these include: 1) the extensive nature of the field, and laboratory data required for application; 2) AET established criteria may not be environmentally conservative (i.e. biological effects can be observed at levels well below the AET for any chemical); 3) the AET approach does not account for interactive effects among chemicals (e.g. additivity and synergisms; even antagonisms may elevate AETs where they do occur, leading to the potential for adverse effects where they do not); 4) AETs assume that a cause and effect relationship exists between bulk chemistry and biological effects; and 5) the potentially confounding effect of co-varying toxicants (measured or unmeasured) in producing false AETs for correlated chemicals.

Points number 1 and 2 stand without comment being necessary. Point number 3 is a potentially fatal flaw since knowledge of the potential toxicological effects of complex mixtures of chemicals such as are commonly found in industrialized estuaries and seaports is very limited. For example, recent studies in our laboratory have indicated that copper influences the bioavailability of cadmium through geochemical and/or biological mechanisms (Rule and Alden, in preparation). Therefore, an AET based criterion established for one contaminant may not be protective in an environment where other interactions with other chemicals occur.

The fourth point is closely related to this issue since studies have shown that concentrations determined by bulk chemical analysis provide very poor predictors of biological availability (see e.g., Lee et al., 1975; Engler, 1978; Neff et al., 1978; Pequegnat et al., 1978; Cross and Sunda, 1979; and others). For example, Rule and Alden (1989) demonstrated that a geochemical fraction (the easily reducible phase) which represents only a small fraction (usually less than 10%) of the total cadmium pool was the best predictor of biological uptake of cadmium from sediments. The majority of the cadmium in estuarine sediments appears to be bound up in the organic-sulfide fraction, which is apparently not biologically available. The interest in the use of acid volatile sulfides to "normalize" the concentrations of metals in sediments (DiToro et al., In Press) may serve to reduce this problem.

The last point concerning the problem of correlation among contaminants was the subject of a series of computer simulation investigations similar in nature to those described above for Triad assessments. It should be noted that additional studies were also conducted to address a variety of related topics:

1) Considering the natural variability in sediment contamination and toxicity, how many samples are needed to reach a stable AET?

2) What are the effects of the incidence of significant toxicity on the number of samples needed to achieve a stable AET? (i.e. How many samples are required to attain a stable AET for various levels of contaminant-related toxicity?).

3) What are the effects of different levels of random mortalities (i.e. toxicity not directly related to the contaminant(s) of concern) on the attainment of a stable AET?

4) What are the effects of the sampling regime (e.g. random, uniform, directed) on the number of samples required to attain a stable AET?

5) What are the effects of random mortalities on the incidence and persistence of "false alarms" (i.e. AETs "set" by toxicity not related to the contaminant(s) of concern)?

Although much information was generated on each of these topics, it is beyond the scope of the present report to discuss these findings. Furthermore, the results of the "correlation" study tend to override the significance of these other topics, so the following discussion will be confined to that issue.

A bootstrap approach was employed to address the "effects of correlation" issue. The data base employed in the study was collected over a several year period in the late 1970s and early 1980s as part of a U.S. Army Corps of Engineers and NOAA funded five-year dredged material assessment program (Alden et al., 1981; Rule, 1986). The particular data sets employed included data on the concentration of nine metals in sediments collected at 52 sites, with four to six sites per site located as cross-channel transects. Thus, a data set representing approximately 300 samples collected and analyzed by Dr. Joseph Rule (Department of Geological Sciences, ODU) from all of the major navigational channels of the Port of Hampton Roads, the channels of the lower Chesapeake Bay, and the nearshore Atlantic waters in the vicinity of a proposed dredged material disposal site formed the data pool from which bootstrap simulations were produced.

The scenario developed for the correlation study involved the following question:

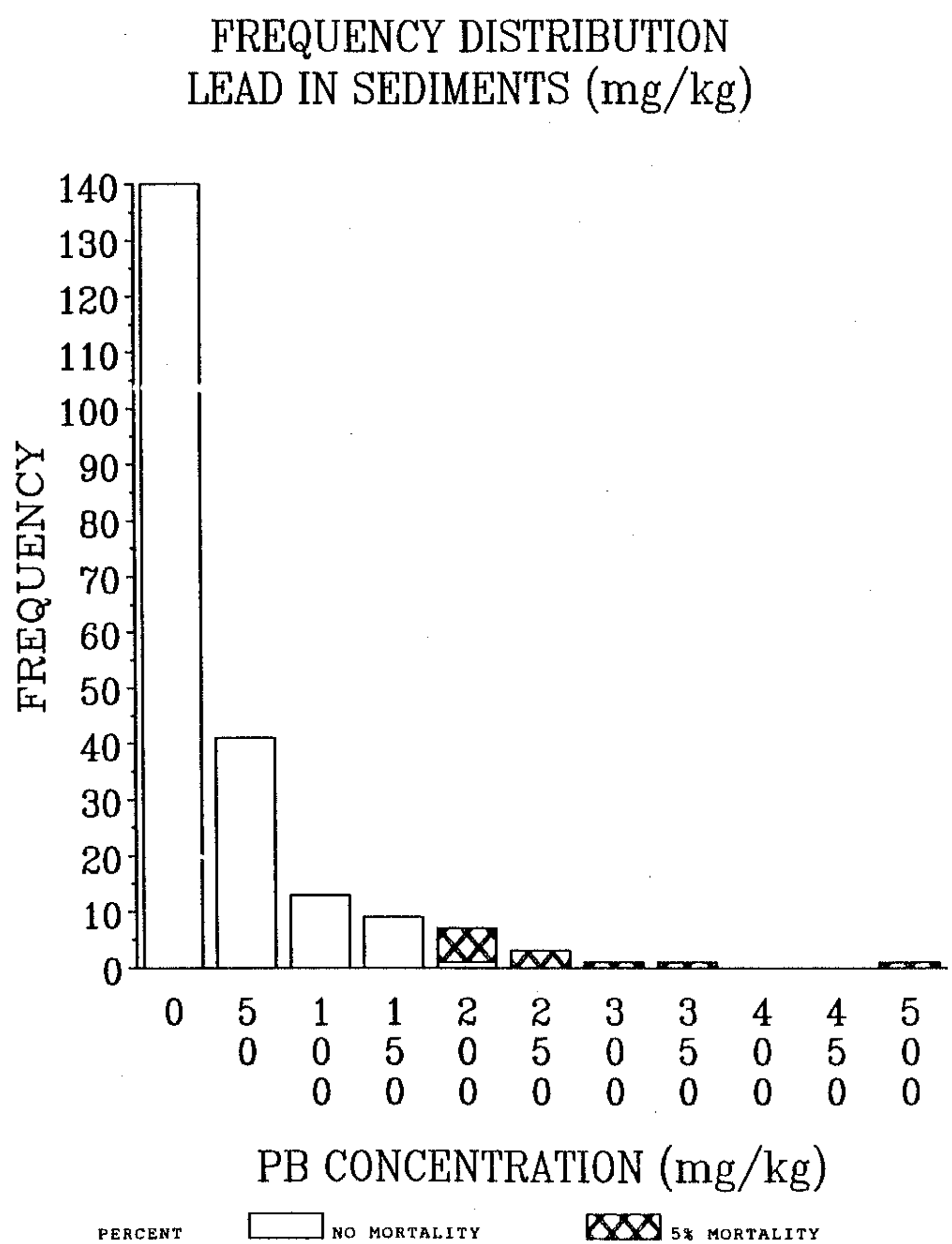
> If only a single contaminant in the samples was responsible for toxicity (i.e. only one chemical had a "valid" AET), how would its correlation to other metals in the samples affect the incidence and persistence of "false" AETs for these (hypothetically) non-toxic metals?

The metal selected as the "true" toxicant was lead. This metal is a known toxicant which is found in high concentrations in certain of the regions of the Elizabeth River. It also displays a range of correlations with other metals, from r=0.42 for manganese to r=0.96 for copper. Lead concentrations in the data base also display the sort of skewed frequency distribution that one would expect for a contaminant which occurs in low concentrations for most of the ambient conditions, but which is found in high concentrations at a few of the highly polluted sites (Fig. 85).

Preliminary studies indicated that the number of samples required to establish a valid AET stabilized at a level less than 5% incidence of toxicity. Therefore, the top 5% of the lead distributions were considered (i.e. lead concentrations greater than 180 mg/kg, Fig. 85) to be toxic. Ironically, the AET level actually calculated at a later date with toxicity data from the Port proved to be very near to this level (Table 27). The computer associated "toxicity" only with the samples containing 180 mg/kg of lead or more. Since the correlations exhibited by the eight other metals to lead did not cover the full range of possible correlations, a simulation routine (Alden, 1984) was employed to produce new "metals" that expanded and filled in this range: simulated metals data with correlations to lead of r = 0, 0.05, 0.10, 0.20, 0.40, 0.50, 0.60, 0.90, and 0.95. These "metals" were simulated to display the overall statistical characteristics (mean, standard deviation, frequency distribution) of copper data, except for the correlation to lead which was varied as indicated. The computer randomly selected samples one at a time and recorded the establishment of AET "levels" as the sample data base grew. Each "run" involved the accumulation of 300 samples (size of original data base). The runs were repeated 20 times for each metal.

Figures 86-88 present visual representations of the simulations for lead, manganese, and copper, respectively. Each

Figure 85. Frequency distribution of lead in sediments collected in 1979 from the Elizabeth River, Hampton Roads Harbor lower Chesapeake Bay channels, and coastal Atlantic waters off the mouth of the Bay. The cross-hatched portion of the bars represents the top 5% of all concentrations which were hypothetically coded as "toxic" (i.e. associated with bioassay mortalities) for the simulation investigation.



FREQUENCY DISTRIBUTION
LEAD IN SEDIMENTS (mg/kg)
FREQUENCY
140
130
120
110
100
90
80
70
60
50
40
30
20
10
0
0
50
100
150
200
250
300
350
400
450
500
PB CONCENTRATION (mg/kg)
PERCENT
NO MORTALITY
5% MORTALITY

Figure 86. Representation of 20 simulation runs for lead in sediments. Each run is represented by a line of a different pattern. As each run progresses (samples are "collected" and evaluated), the lines rise above the x-axis when an incipient AET is established and drop to a lower level when the AET estimate is "refined", or to zero if the AET is proven "false".

LEAD IN SEDIMENTS vs. NUMBER OF SAMPLES COLLECTED
AET ASSESSMENT, LEAD MORTALITIES (5%), RANDOM SEARCH

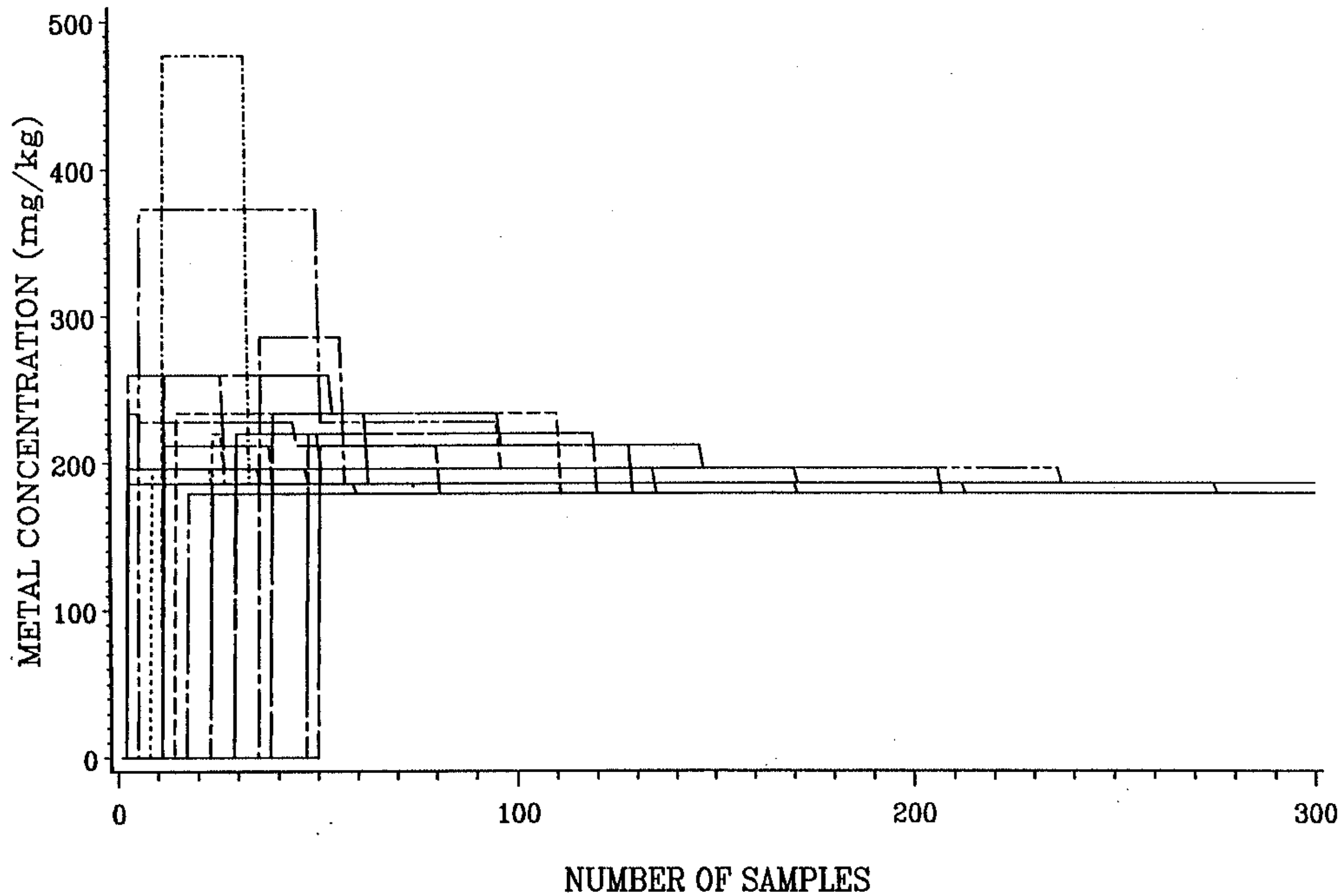


Figure 87. Representation of 20 simulation runs for manganese in sediments. Each run is represented by a line of a different pattern (see Figure 86 and the text for more explanation).

MANGANESE IN SEDIMENTS vs. NUMBER OF SAMPLES COLLECTED
AET ASSESSMENT, LEAD MORTALITIES (5%), RANDOM SEARCH
MANGANESE–LEAD CORRELATION=0.42

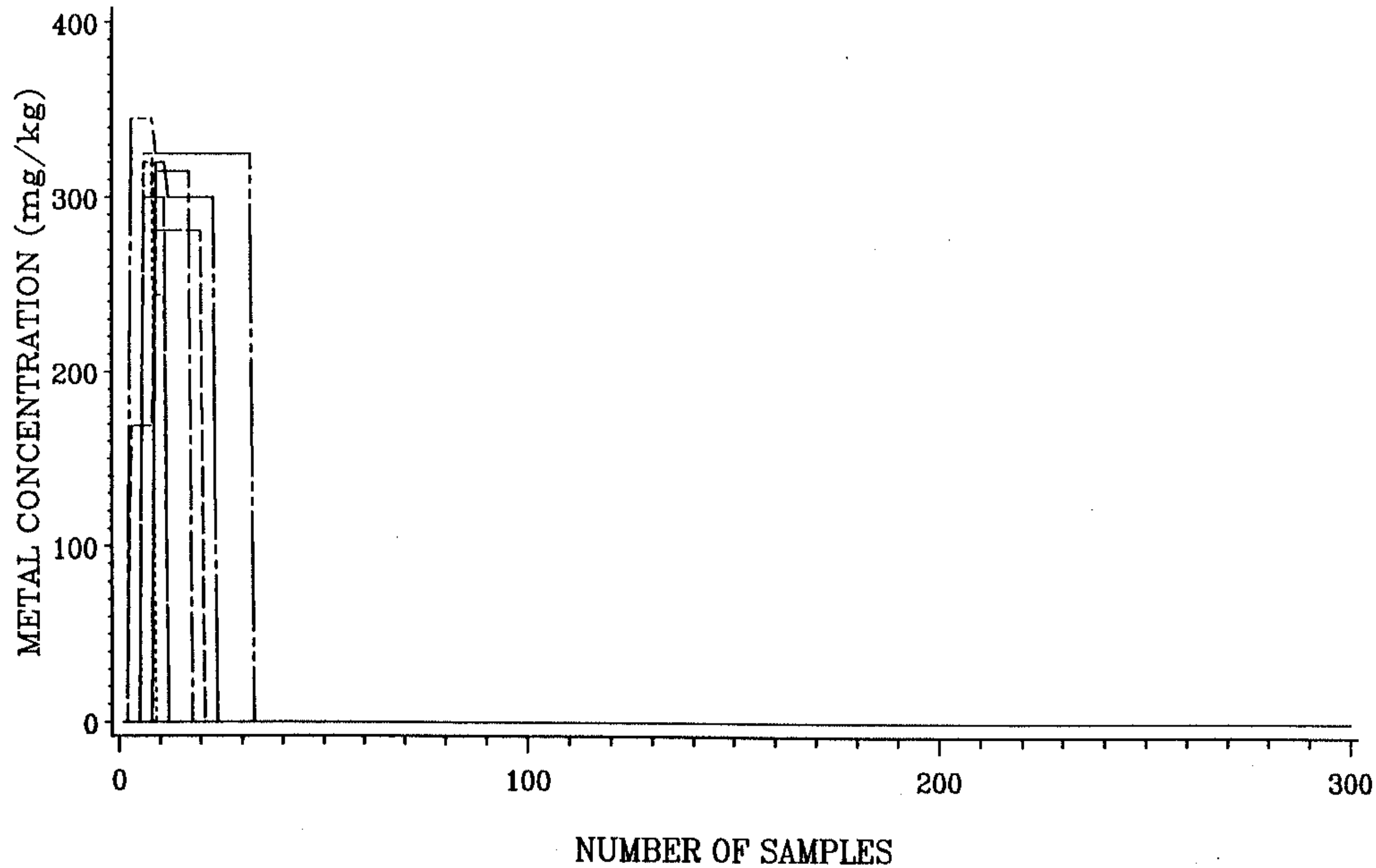
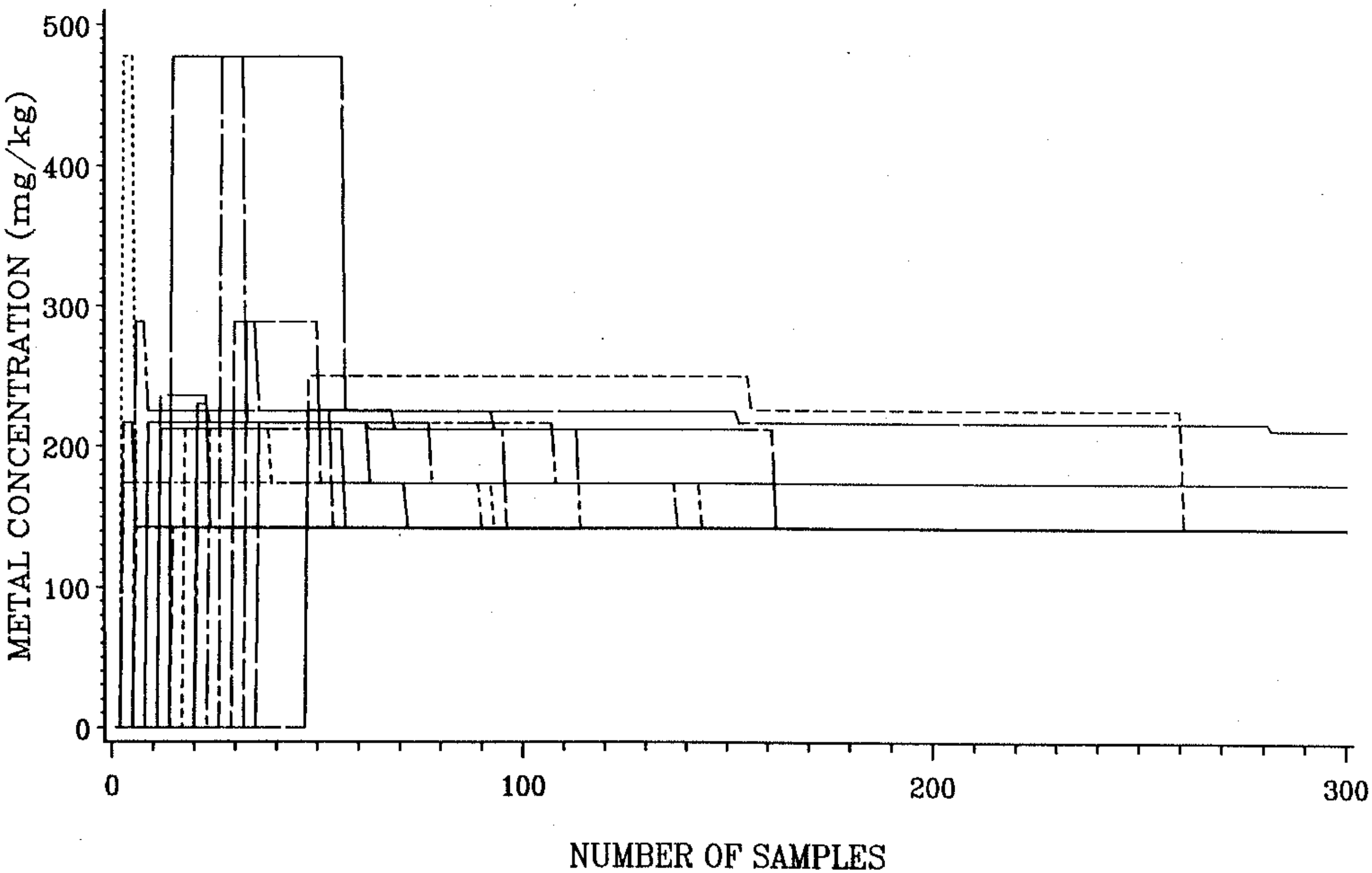


Figure 88. Representation of 20 simulation runs for copper in sediments. Each run is represented by a line of a different pattern (see Figure 86 and the text for more explanation).

COPPER IN SEDIMENTS vs. NUMBER OF SAMPLES COLLECTED
AET ASSESSMENT, LEAD MORTALITIES (5%), RANDOM SEARCH
COPPER–LEAD CORRELATION=0.96



"run" is represented by lines of different patterns (e.g. solid, dotted, dashed, etc.) and the lines rise above the x-axis whenever an incipient AET is established and drop to a lower level when the AET estimate is "refined", or to zero if the AET is proven "false" (i.e. a higher concentration of the metal is shown to be non-toxic). The "valid" AET for lead tends to be approached or reached within 50 samples for most of the runs. On the other hand, manganese displays a few "false alarms" early in the simulations, but these are detected fairly quickly, usually within the first 20-30 samples. However, copper behaves much like lead, even though it is considered "non-toxic" for the purposes of the simulations. An AET level is set and stabilized for each run, usually within the first 50 samples "collected".

Figures 89-91 summarize the data from all simulations. Obviously the incidence of "false alarms" (false AETs) rises fairly dramatically above r=0.50 and are almost certain above r=0.70 (Fig. 89). Also, the number of false AETs produced within 300 samples also rises above a correlation of r=0.50 (Fig. 90). The rise is not as dramatic as it is for the percent incidence because many of the false AETs for metals with high correlations to lead are "fixed" the first time observed. Perhaps the most disturbing information is presented in Figure 91, which indicates that false AETs established for metals with correlations above r=0.6 have very little probability of being detected as being false within a data base as large as 300 samples.

The implications of these findings are quite significant to

Figure 89. Percent incidence of false AETs (out of 20 runs) versus the correlation of "nontoxic" metals with lead, the hypothetical toxicant.

PERCENT INCIDENCE OF FALSE AETs vs CORRELATION BETWEEN "NONTOXIC" METALS & LEAD (PB MORTALITIES=5%)

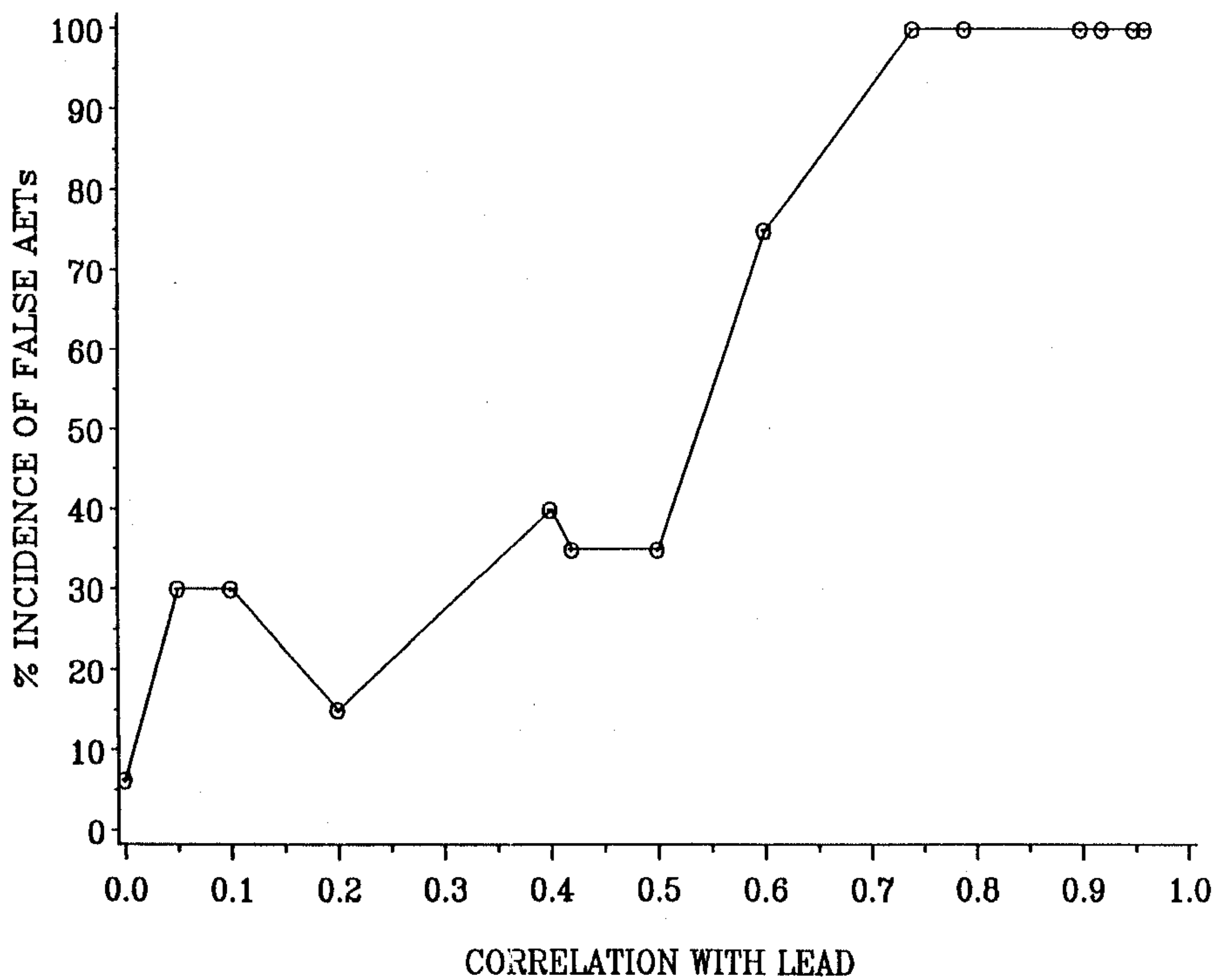


Figure 90. Mean number (± 2 std. errors) of "false" AETs per run versus the correlation of "nontoxic" metals with lead.

NUMBER OF FALSE AETs per RUN (300 SAMPLES) vs CORRELATION BETWEEN "NONTOXIC" METALS & LEAD (PB MORTALITIES=5%)

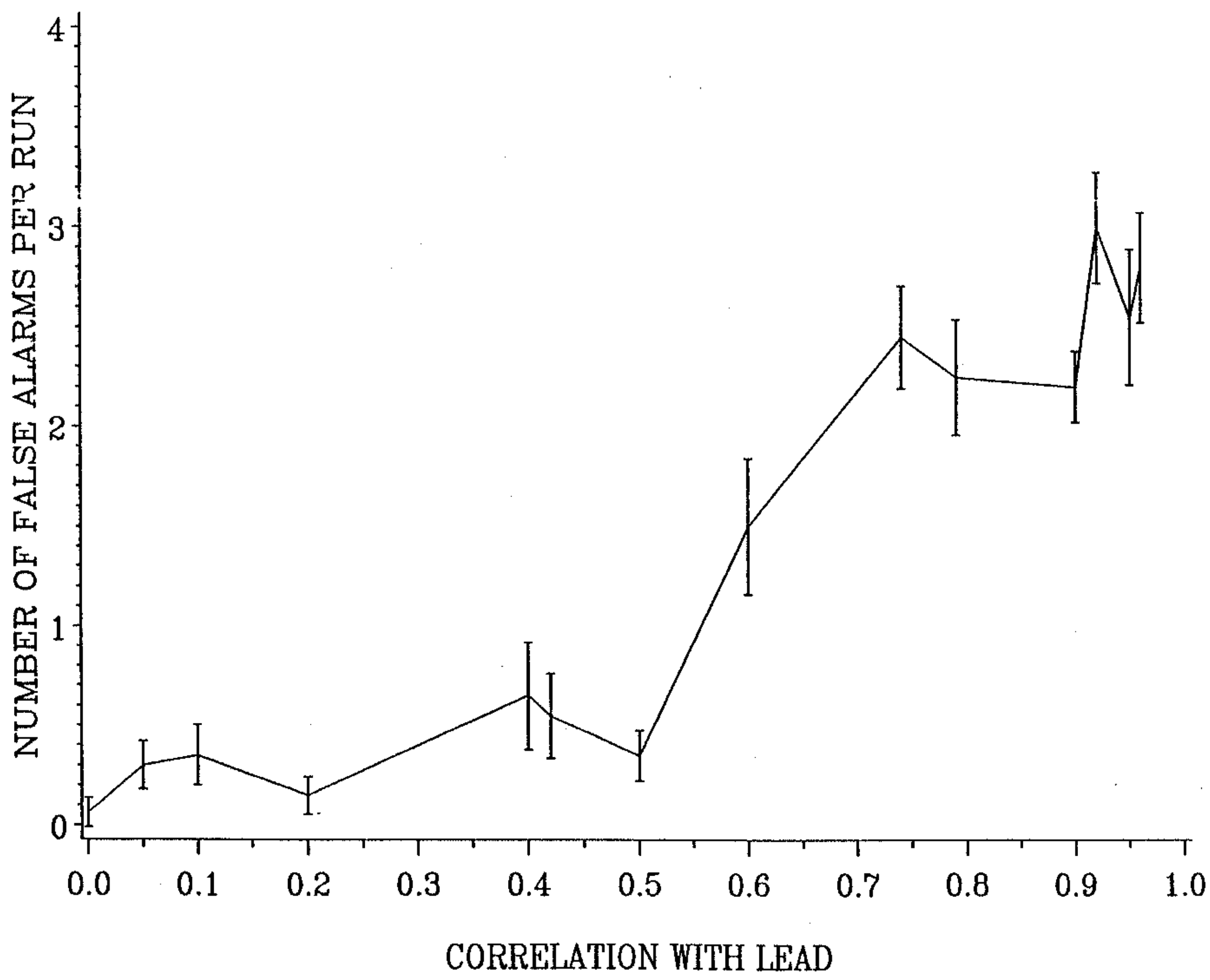
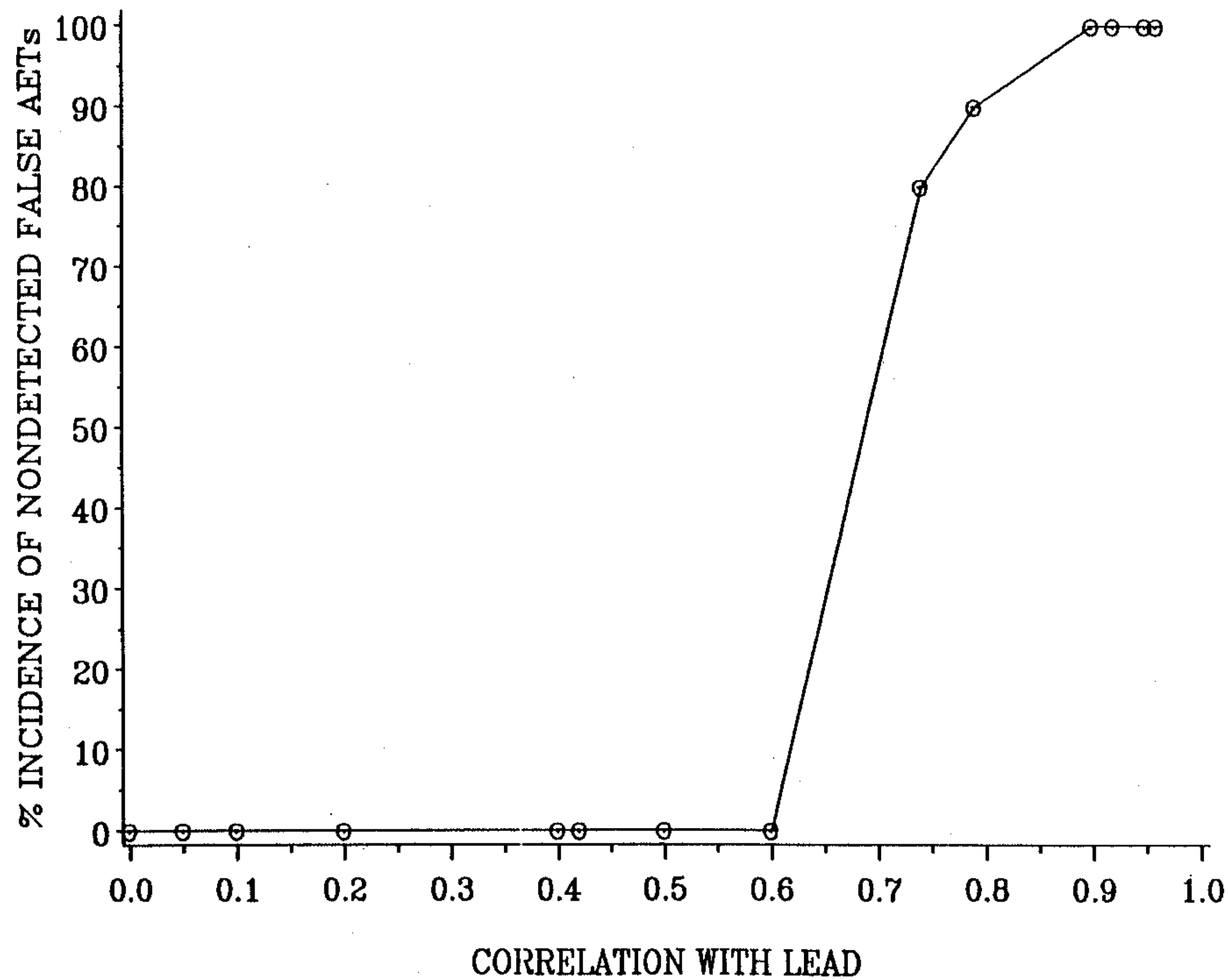


Figure 91. Percent incidence of invalid AETs that are not detected as "false" (within 300 samples analyzed) versus the correlation of "nontoxic" metals with lead.

PERCENT INCIDENCE OF NONDETECTED FALSE AETs vs CORRELATION BETWEEN "NONTOXIC" METALS & LEAD (PB MORTALITIES=5%)



ultimate use of the AET approach. Most contaminated estuarine sediments are the product of multiple sources of pollution. Therefore, it is highly likely that sediments contain a large number of contaminants, many of which are highly correlated to each other. Of course the correlations quite likely depend, to a large degree, on the mix of sources in any given area. Therefore, the complications observed for correlation effects in this study could confound any attempt at establishing valid AET-based sediment quality criteria. Known and unknown correlations between contaminants, both measured and unmeasured, would confuse the establishment of valid AETs. Furthermore, AETs established in one estuarine ecosystem, with a given "correlation structure" among contaminants, may not be validly applied (for regulatory compliance or remediation purposes) to another system, where the correlations between contaminants (known and unknown) may be quite different. To extend the scenario used for the simulations as an example of this point, an AET established for copper in Hampton Roads may be due to lead toxicity, but it may also be protective for the environment since the two contaminants are highly correlated. In other words, if sediment copper concentrations were "cleaned up", lead might also be controlled, assuming common sources as well as common responses of the metals to control strategies. However, to utilize the copper AET in another system, where the correlation to lead may be quite different, might prove fruitless (and costly) in controlling sediment toxicity.

Considering the complex nature of sediments from industrialized estuaries, the high probability of establishing "false" AETs that cannot be detected as false would appear to be insurmountable, especially if the goal is the establishment of defensible sediment quality criteria. Rather, an AET should be seen as a "first cut" level of concern, which would need to be validated by techniques such as spiked sediment toxicity tests (utilizing a variety of sediment types, and other environmental conditions) before regulatory actions are formulated. However, using AETs for this "screening" role may not be extremely cost effective. The considerable cost of developing the magnitude of data base required for establishment of AETs might be better applied to widespread monitoring of sediments by sediment toxicity tests to define "hot spots" which can be further examined and managed.

The philosophy used by many of the states in NPDES monitoring of effluents has shifted focus from chemical analysis to biomonitoring. Any effluent shown to be toxic can be subjected to a Toxicity Reduction Evaluation/Toxicity Identification Evaluation (TRE/TIE), whereby the toxicants are identified and (or) controlled. A similar philosophy could be applied to sediment quality concerns. Letting the responses of biota, either in the laboratory or in <u>in situ</u> communities, define sediments that are of poor enough quality to merit further studies would appear to represent an effective screening strategy. Extensive chemical surveys, toxicological investigations (i.e. "Triad" studies), source identification studies, and clean-up strategies could then follow in a TRE/TIE-like scenario. In other words, the best approach to sediment quality assessments may be to let biological systems integrate the complex nature of the contaminants and indicate whether the sediments are of poor enough quality to detrimentally impact on living resources.

Equilibrium Partitioning

Application of EP Approach to Elizabeth River Data

Another approach to sediment quality assessments is the "Equilibrium Partitioning" (EP) approach to the calculation of sediment quality criteria for non-polar organic compounds (Tetra Tech, 1986; EPA, 1989a). The general approach to this method is detailed by Tetra Tech (1986) and EPA (1989a), but basically involves a numerical model which allows the calculation of a sediment quality criterion from the appropriate water quality criterion:

$$C_s = K_{oc} \; C_w \qquad [1]$$

where C_w is the water quality criterion for the contaminant of concern, C_s is the calculated sediment quality criterion (expressed as a concentration that has been "normalized" to the organic carbon component of the sediments) and K_{oc} is the organic carbon/water partition coefficient. The K_{oc} value can be estimated from the K_{ow}

(octanol/water partition coefficient) of the contaminant and an empirically determined regression relationship:

$$\text{Log } K_{oc} = a \log K_{ow} + b \qquad [2]$$

where a and b are constants (e.g. a=0.989 and b=0.3456 in one of the most commonly used equations; see equation 8 in section entitled **Sediment Fluxes of Organic Contaminants**). Organic carbon-normalized sediment concentrations of the contaminants of concern from each site are then compared against these sediment quality criteria.

Unfortunately, among the non-polar organic contaminants observed in the sediments (Greaves, 1990), only three (naphthalene, acenaphthene, and fluoranthene) have previously established water quality criteria. Figures 92-94 present the carbon-normalized concentrations of these contaminants along with a reference line representing the value of the calculated sediment criterion. The most obvious conclusion that can be drawn from this type of presentation is that the contaminants in the samples from the Phase I ERLTMP study never approached within an order of magnitude of the calculated sediment quality criteria. Furthermore, even the <u>maximum</u> concentrations of these contaminants that have been reported in any of the previous studies of the Elizabeth River (designated "maximum") were not within the same order of magnitude as the calculated criteria. Estimates of sediment quality criteria based upon <u>chronic</u> water quality criteria were not less than 1/2-1/3 of the values calculated for acute criteria, so even these

Figure 92. Organic carbon-normalized concentrations of naphthalene at the reference and monitoring sites compared to the calculated sediment quality criterion.

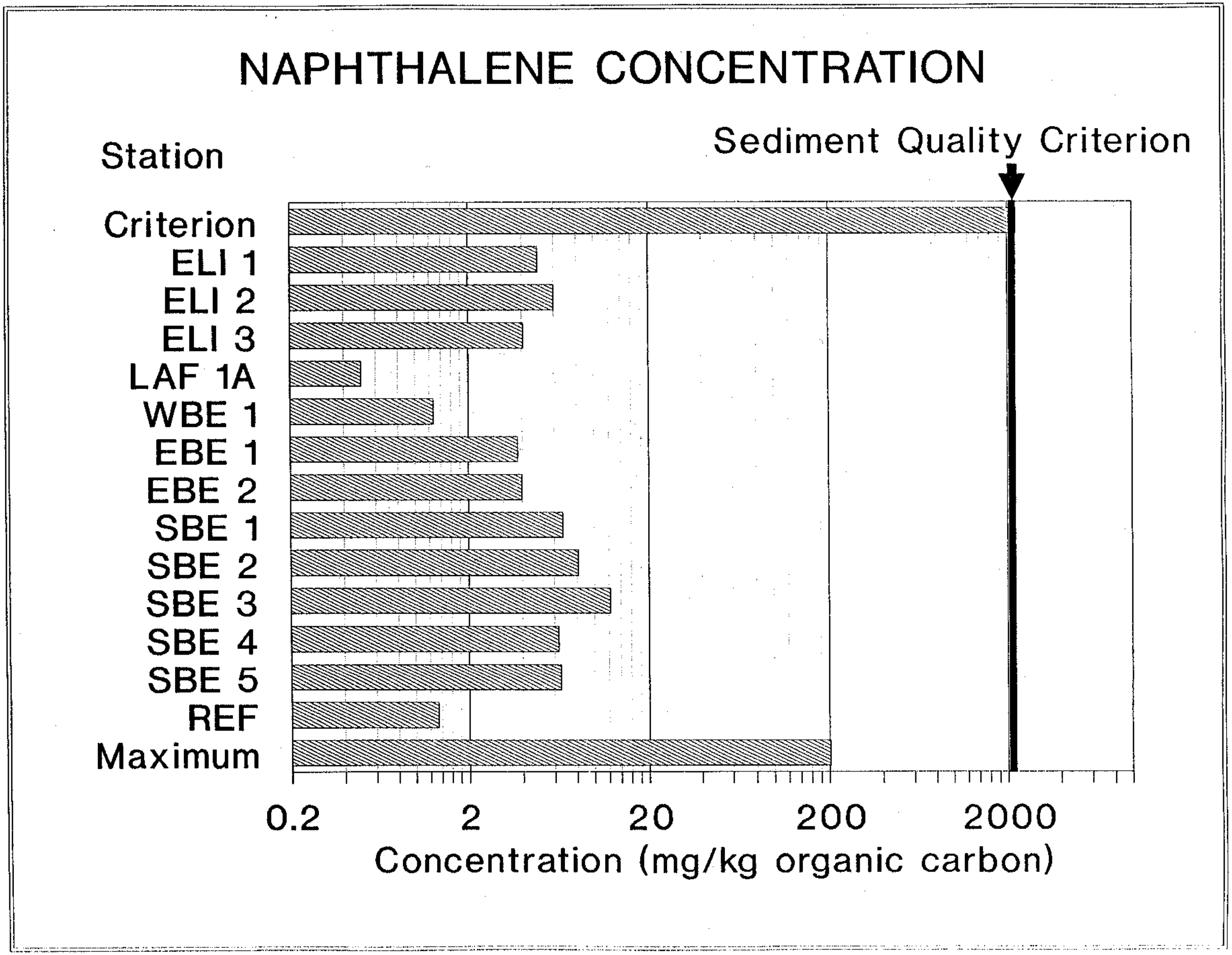

NAPHTHALENE CONCENTRATION
Sediment Quality Criterion
Station
Criterion
ELI 1
ELI 2
ELI 3
LAF 1A
WBE 1
EBE 1
EBE 2
SBE 1
SBE 2
SBE 3
SBE 4
SBE 5
REF
Maximum
0.2
2
20
200
2000
Concentration (mg/kg organic carbon)

Figure 93. Organic carbon-normalized concentrations of acenaphthene at the reference and monitoring sites compared to the calculated sediment quality criterion.

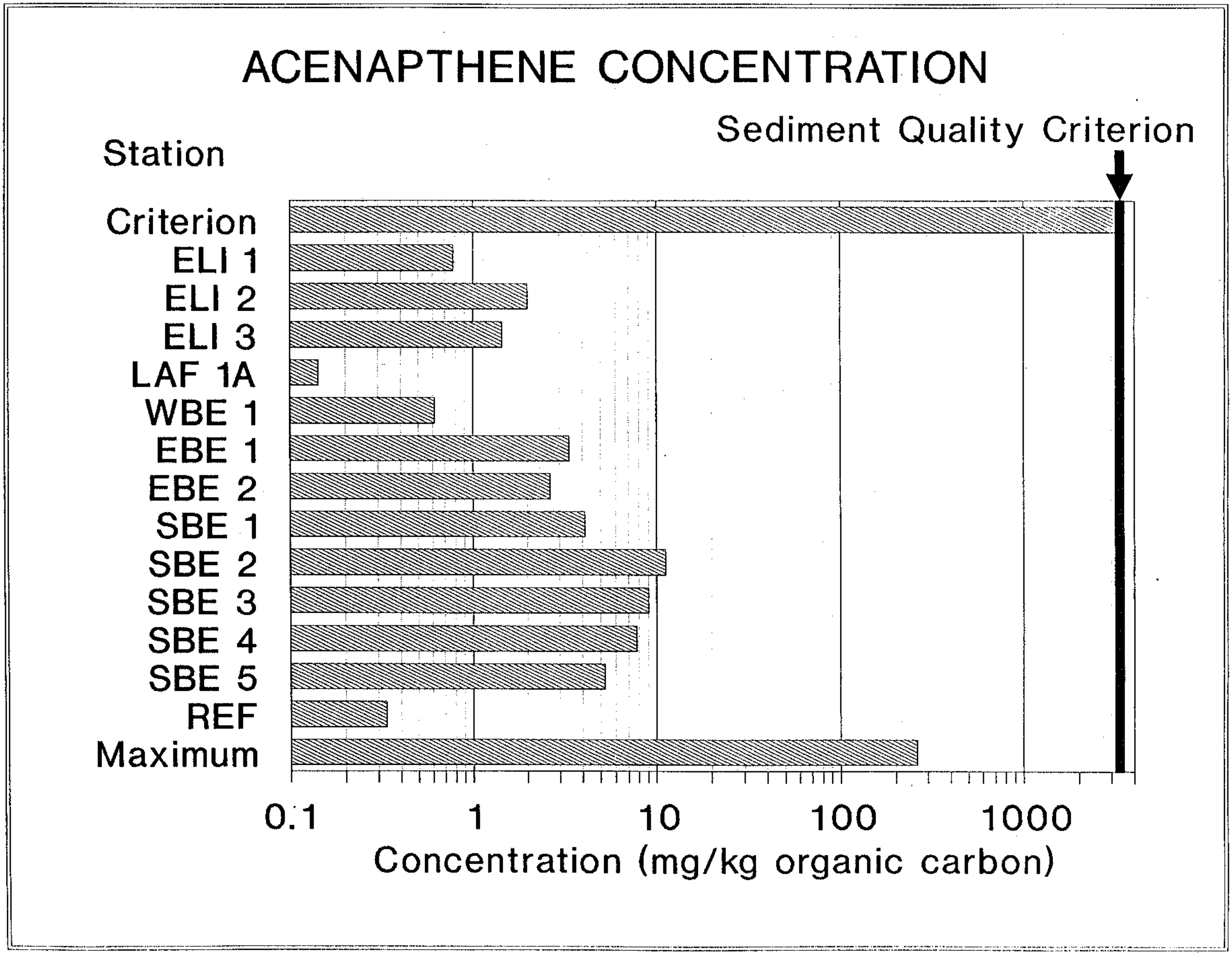
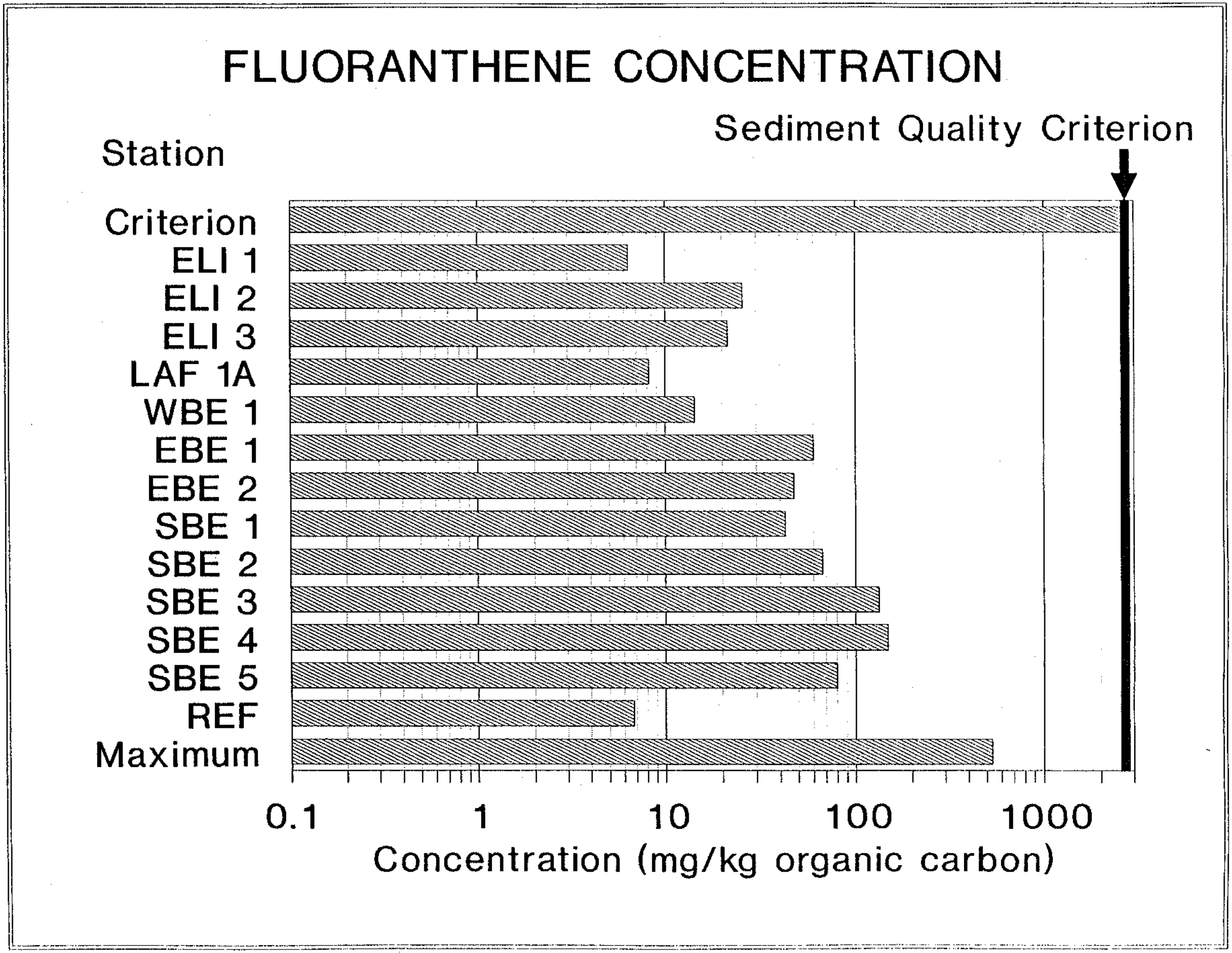

ACENAPTHENE CONCENTRATION
Sediment Quality Criterion
Station
Criterion
ELI 1
ELI 2
ELI 3
LAF 1A
WBE 1
EBE 1
EBE 2
SBE 1
SBE 2
SBE 3
SBE 4
SBE 5
REF
Maximum
0.1
1
10
100
1000
Concentration (mg/kg organic carbon)

Figure 94. Organic carbon-normalized concentrations of fluoranthene at the reference and monitoring sites compared to the calculated sediment quality criterion.



FLUORANTHENE CONCENTRATION
Sediment Quality Criterion
Station
Criterion
ELI 1
ELI 2
ELI 3
LAF 1A
WBE 1
EBE 1
EBE 2
SBE 1
SBE 2
SBE 3
SBE 4
SBE 5
REF
Maximum
0.1
1
10
100
1000
Concentration (mg/kg organic carbon)

would not have been approached. The levels of organic contaminants that have been observed in the sediments from certain areas of the Elizabeth River have been shown to be highly toxic (Alden and Young, 1982; Hargis et al., 1984; Alden and Butt, 1988, Alden et al., 1988; Alden and Butt, 1989), so the fact that criteria for individual PNAs (particularly fluoranthene, the dominant PNA) have not been exceeded in any of the previous studies brings their sensitivities into serious question.

Assessment of the EP Approach

The limitations of the EP approach to the establishment of sediment quality criteria have been previously discussed (Tetra Tech, 1986). Basically, the limitations involve the following issues:

1) It is applicable only to non-polar organic compounds (although preliminary explorations are being made for normalizing metals to acid volatile sulfides; Ditoro et al., 1990), and very few have water quality criteria established.
2) There is uncertainty as to the route of biological uptake of contaminants in sediments, yet the method assumes it is through water (pore water or near-bottom water).
3) The method ignores the importance of interactions with other environmental factors:
   a) Refractory organic matter may delay or prevent

steady-state conditions;

b) Dissolved or colloidal organic matter may influence the magnitude of partition coefficients relative to the laboratory derived values which have been based upon experiments using pure water;

c) Many previous K_{oc}-K_{ow} relationships from the literature are dependent on rather low sediment to water volume ratios which may not be equivalent to <u>in situ</u> conditions which the EP approach attempts to model;

d) Other factors such as temperature, salinity and particle size may influence relationships.

4) Like all single chemical criteria/standards, the values produced by the EP approach do not take into account interactions (additivity, synergisms, antagonisms) which may influence toxicity of the complex mixture of contaminants characteristically found in sediments from industrialized sediments.

The results observed from the application of the EP approach to Elizabeth River sediment contaminants data suggest that it may not be as sensitive as it should to be environmentally protective. One must pose the question: If EP-based criteria for chemicals such as fluoranthene are not violated in the worst areas of the Elizabeth River (which have among the highest sediment concentrations of this compound in the world), what environments will they protect?

As previously discussed, the sediments from this system have been shown to contain high concentrations of PNAs which apparently produce a variety of toxic effects. Either the toxicity is due to a combination of PNAs (Issue #4) or the model is too simplistic to adequately describe the potential toxicity of the sediments (Issues 2 & 3).

The EP approach for establishing a fluoranthene sediment quality criterion has been verified to predict sediment toxicity under certain specific conditions (sandy sediments with low carbon content; see Swartz et al., 1990). However, the conditions of this test were quite different from those found in the Elizabeth River or in many other industrial estuaries. The sediment was sandy, in contrast to the silt-clay that dominates the Elizabeth River, and the TOC levels were 1-2 orders of magnitude lower than those observed for the Elizabeth River. Such differences may add such a degree of complexity to the picture that the EP models may break down. In fact, Swartz et al. (1990) are very careful to point out that their results are for sandy sediments with low carbon and that they "should not be extrapolated uncritically to other taxa, chemicals or sediment types."

There is a considerable potential that high levels of organic contaminants dissolved or emulsified in pore water of Elizabeth River sediments* may act as carriers to "extract" toxicants off the particles, producing far lower K_{oc} values and greater toxicity than predicted by the EP equations (Issue #3b). If the approach is to be universally applied, empirical data from "real-world" samples would have to be examined to determine the magnitude of deviations of pore water concentrations of organic contaminants from the predicted C_w values.

* Liquid creosote and oil products can be observed to ooze from sediment samples collected from a number of areas of the Elizabeth River.

MANAGEMENT IMPLICATIONS OF ELIZABETH RIVER SEDIMENT STUDIES

Contaminants found in estuarine sediments represent a potential risk to human health and the living resources. This reservoir may prevent environmental managers from attaining and maintaining the water quality goals of the CWA in the Elizabeth River due to in situ effects or due to the possibility of reintroduction of pollutants into the water column. Section 115 of the CWA directs the identification of areas containing in-place pollutants, with special emphasis on toxic substances, and mandates a policy for subsequent removal and disposal from critical port and harbor areas. A decision to remove and dispose of contaminated material presupposes that the sources have been significantly reduced or eliminated and that removal and disposal options will not create greater environmental risk or damage.

The routes of exposure and hazard potential to humans and the living resources which may be associated with contaminants in sediments, as well as the influence the contaminants may have on estuarine surface water quality, are not completely understood. Although not well documented, the fate of these substances in the estuarine and marine environment should also be considered when evaluating the potential for unacceptable adverse environmental impacts.

Some of the information needed to address the problem of in-place contaminants is defined under Section 117 of the CWA:

" (a)(3) determine the impact of sediment deposition in the Bay and identify the sources, rates, routes, and distribution patterns of such sediment deposition; and

(a)(4) determine the impact of natural and man-induced environmental changes on the living resources of the Bay and the relationships among such changes, with particular emphasis placed on the impact of pollutant loadings of nutrients, chlorine, acid precipitation, dissolved oxygen, and toxic pollutants, including organic chemicals and heavy metals, and with special attention given to the impact of such changes on striped bass."

There is no single comprehensive study, including the present effort, which provides the answer to all questions concerning the source, fate and effects of contaminants in the Elizabeth River. The major obstacle facing resource managers is the lack of access to a useful data base which contains all environmental data for the Elizabeth River and would allow integration and analysis in a manner which supports environmental protection guidelines and regulations (EPA, 1986).

Conclusions drawn from the data presented in this study and comparisons to the results of previous studies are useful in supporting management strategies required to address in-place contaminants. The process of restoring the water quality of the River (the goal of the Elizabeth River Initiative) requires the development of a priority ranking system for focusing management attention. The scheme presented here (see Figure 95) relies on a programmed approach beginning with the acquisition of a data base (TASK I.) designed to support management needs.

The data base should allow hazard assessments (TASK II.) and exposure assessments (TASK III.) through access to site specific data from studies conducted in the Elizabeth River. These data will be used for assigning a ranking of priority "hot spots" (TASK IV.) for immediate attention. Chemical-specific "hot spots" are identified, but not ranked, in this report. Preliminary ranking of sites using chemical-specific concentrations and toxicity will be presented in the ERLTM/MP (Elizabeth River Long-term Monitoring/Management Program) Phase II study.

Using the available data, the manager would determine the potential for environmental risk or the presence of "damaged" areas (TASK V.) at targeted "hot spots." Environmental risk/damage assessment integrates hazard (dose and endpoint) and exposure (concentration and fate) data with knowledge of the population (species, living resources, humans) to determine the potential for unacceptable adverse impact or the presence of a damaged system. Although quantitative human health risk models are well documented and have regulatory acceptance, no comparable environmental risk or damage model is currently available (Krome, 1989). Qualitative risk assessment evaluations seem to be the only approach available to resource managers at this time. Monitoring programs should continue at all sites until sufficient data indicate the trend is

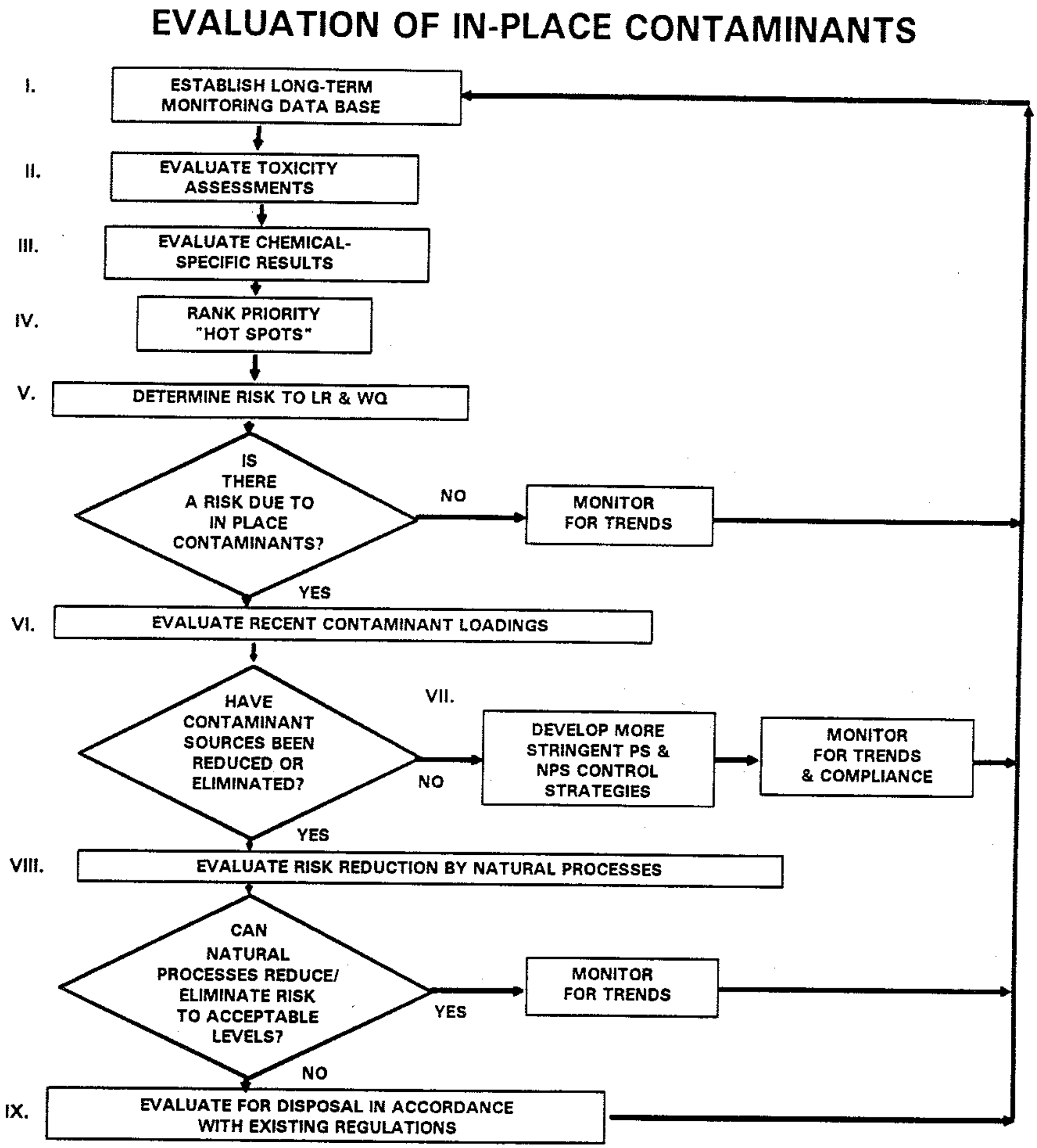


Figure 95. Schematic approach for an environmental resource manager's evaluation of in-place contaminants.

toward meeting CWA goals.

If it is determined that the potential for an unacceptable adverse environmental impact exists from in-place contaminants, it is necessary to determine if the sources are continuing to contribute to the contaminant loadings (TASK VI.). If the results indicate active contaminant sources, the manager must develop and execute more effective pollutant controls (TASK VII.). Thus, the manager must have sufficient evidence that pollution controls or stable use activities will prevent additional contaminant loading prior to considering the option of removal and disposal of in-place contaminants. The results of site specific studies and continued monitoring will determine if and when the controls are effective and management actions are needed.

When source controls have been shown to be effective, the manager should assess whether the magnitude of risk presented by in-place contaminants is persistent (TASK VIII.) thus, warranting removal and disposal. This assessment process should consider sedimentation rates (burial) and fate (flux; persistence; bioaccumulation potential; resuspension: due to dredging, scouring by ship activity, sediment movement during storm events, bioturbation, etc.) when estimating the magnitude of risk. If sedimentation rates or other factors indicate that the hazard and exposure due to contaminated sediments could be reduced in a reasonable period, then long-term monitoring programs should include the site(s) to validate the decision to leave the contaminants in-place.

If dredging is determined to be the only option for reducing or eliminating the risk/damage, the final step (TASK IX.) in this process would be an ecological evaluation of the contaminated sediments. Existing federal and state regulations provide for selecting sediment removal, transport and disposal techniques which will reduce the potential for additional contamination of the water during dredging activities.

With sufficient information, the manager can perform environmental risk/damage assessments and evaluate dredging options or requests for dredging activities in areas containing contaminated sediments. The approaches to evaluating dredge material for disposal and dredging technique strategies is well documented in the regulatory literature. The following findings and implications support, but do not provide all of the information required to formulate fully justified management options. The ERLTM/MP Phase II study will attempt to develop more fully the management options to address in-place contaminants.

In the following section, the FINDINGS/OBSERVATIONS pertinent to each task are identified by letters, while the management IMPLICATIONS are numbered in sequence (1-45) for reference purposes.

TASK I. ESTABLISH LONG-TERM MONITORING DATA BASE

A. FINDINGS/OBSERVATIONS: The procedure for comparing data generated by this study with results of previous studies conducted in the Elizabeth River was cumbersome and trend assessments were limited because the majority of existing data were not available to the authors.

IMPLICATIONS

1. Access to existing data is limited to the results of previous studies conducted by these authors, information in "gray" literature with limited circulation, and a few published articles.

2. It is impossible to know with certainty that comparisons of data generated from samples collected at specific sites in this study are representative of the same sites examined by previous studies.

B. FINDINGS/OBSERVATIONS: The major obstacle facing resource managers when attempting to conduct site-specific evaluations is the lack of access to a useful data base on environmental conditions in the Elizabeth River.

IMPLICATIONS

3. The development of any data base should ensure that it

meets the needs of the users and will be easily accessible. Although it is difficult to design a single data base to meet the requirements of multiple users and programs, it is possible, providing the data base is designed to address specific problems within well defined areas. The problem of sediment contaminants in the Elizabeth River system warrants such a data base. Following guidelines developed by the Computer Sciences Corporation for the Chesapeake Bay Program (CBP) (USEPA, 1989b), a data base can be developed to meet the goals of the CBP and to support management actions in the River.

4. It is essential that the data base allow entry and retrieval of information suitable for conducting site-specific evaluations. A limitation of the Chesapeake Bay Toxics Initiative Data Base (CBTIDB) is the difficulty in retrieving information concerning sites identified by location, either by VWCB River Mile, latitude/longitude, or Water Body Segment. Retrieval of data from the CBTIDB for use in statistical analyses is problematic at best. Additionally, cumulative frequency distributions of suspected toxic substances which are produced by the VWCB are not available for individual sites or specific segments such as in the Elizabeth River.

5. Information from environmental studies (i.e. geophysical, hydrological, ambient toxicity and chemical-specific data)

should be assembled and incorporated into a data base, preferably a GIS-type data base, which will allow resource managers the ability to examine spatial and temporal patterns and perform trend assessments of substances in water and sediments of the Elizabeth River.

6. The USGS Elizabeth River GIS Project (VWCB, 1988) may provide a useful data base either as a source of additional information or as the primary data base. The previously mentioned criteria should be used in determining whether it is the tool needed by the resource managers.

7. A data base composed of information should be developed which will support the development of sediment quality criteria (SQC) for toxicants of concern. Although current approaches to the development of SQC do have limitations (see IMPLICATIONS 26), data bases should be available for SQC calculations, should they be established by the EPA in the future.

C. FINDINGS/OBSERVATIONS: Comparisons of data from a variety of studies may not be valid due to differences in sample collection and analysis methods (e.g. differences in detection limits), location of sampled sites, seasons in which the material is collected and, most importantly, due to the quality of the data reported.

IMPLICATIONS

8. A quality assurance screen should be developed and applied to all data which may be used to support regulatory activities.

9. Guidelines for a data quality assurance program are detailed in EPA, 1989b and provide procedures to ensure the data are "...secure, valid, complete, and of known quality." Other criteria of equal importance include whether the data were representative of the site or area sampled, and whether the appropriate analytical methods were used. These specific requirements must be addressed prior to the development of a data base because of the potential for drawing invalid interpretations or conclusions from the stored information.

D. FINDINGS/OBSERVATIONS: Relative concentrations of contaminants may aid in the identification of the worst sites within a system, but this information does not yield insight into the magnitude of impact unless comparisons are made to representative, uncontaminated reference sites.

IMPLICATIONS

10. The magnitude of differences can be assessed when environmental observations from the system in question are compared to a reference site, where it has been demonstrated that no significant impact exists. This approach could provide reasonable goals for restoring the site to natural conditions.

However, the reduction/ elimination of ambient toxicity and effects on living resources will always be the primary goals.

11. Criteria for selecting and characterizing reference sites must include, but not be limited to; 1) geochemical and geophysical similarity to impacted sites: 2) composition of living resources equivalent to impacted sites; and 3) the site is not expected to be impacted by industrial or urban activities in the near future.

12. All data from previous studies should be compared to those from appropriate reference sites to provide a method for ranking priority sites or areas based on magnitude of difference, using ambient toxicity, bioaccumulative potential, and *in situ* effects as the most important criteria (see IMPLICATIONS 13, 16, 22).

TASK II. EVALUATE TOXICITY ASSESSMENTS (HAZARDS)

FINDINGS/OBSERVATIONS: The focus of the present report was not in producing toxicity assessments. However, several findings are relevant to this topic:

A. Solid phase bioassays employing *Mysidopsis bahia*, *Palaemonetes pugio* and *Cyprinodon variegatus* for sites evaluated during ERLTM Phase I did not indicate acute toxicity;

B. _In situ_ effects on benthic communities (biomass and diversity) were indicated in the triad assessment of these sites (see FINDINGS/OBSERVATIONS IV.A. AND IMPLICATIONS 21);

C. The concentrations of at least one toxic contaminant (mercury) exceed the "action level" (300 ppb in sediments) that is designed to trigger special studies related to potential human health effects. While no specific action levels exist for them, a number of other known toxicants/mutagens such as dibenzofuran, and high molecular weight PNAs (benzofluoranthene and benzo(a)pyrene, etc.) were observed in sediments throughout the system.

IMPLICATIONS

13. The low levels of acute toxicity in some areas previously shown to be highly toxic indicates that these areas may be improving in overall sediment quality. However, the ERLTM Phase I sites did not cover some of the regions that were seen to be most toxic in previous investigations. Furthermore, the indicated _in situ_ effects and elevated sediment contaminant levels, as well as observations from other investigations that chronic effects on living resources continue in the Elizabeth River suggest that more widespread and more sensitive biomonitoring protocols be implemented to fulfill this task. The on-going ERLTM/MP provides a system-wide assessment taking this approach. However, more extensive studies (possibly

incorporated as industrial permit monitoring requirements, see IMPLICATIONS 27 and 39-41) may be required to delineate hazard risk and/or to track trends on a site-specific basis (particularly in "hot spots").

14. The presence of known biological "hazards" (such as mercury and organic contaminants) in the sediment in elevated amounts suggests that further studies be conducted to allow the potential risk to be assessed. The ERLTM/MP Phase II study will provide data on bioaccumulation of certain contaminants in crab tissues, but other routes of exposure are not being assessed. The widespread distribution of dibenzofuran in sediments of the system suggests that preliminary surveys for the presence and hazard assessment of dioxin in the sediments should be considered.

15. VWCB has established an action level for mercury of 300 ng/g in freshwater sediments (VR680-21-01.10). When this level is exceeded, or when mercury in sediments exceeds historical baseline concentrations, the VWCB is authorized to collect and analyze edible fish tissue from two species of freshwater fish from two trophic levels. There is no estuarine or marine sediment action level for mercury. Since there are no Virginia Department of Health fishing bans or advisories in effect for the Elizabeth River (VWCB, 1990a) and the results of this study indicate sediment mercury concentrations exceed 300 ng/g, it is

suggested that edible fish tissue be collected from the appropriate sites (particularly in the vicinity of Site N) for mercury analysis.

16. Current laboratory toxicity assessment methods may not provide a complete understanding of the potential impact to the living resources resulting from contamination in sediments. Hazard resulting from contaminants in sediments may not be apparent when using existing acute or chronic toxicity methods. The use of guidelines similar to those provided for the ecological evaluation of dredge material for disposal (EPA/ACOE 1979, ACOE 1976) may be required to acquire evidence of the hazard. The guidance manuals provide a structured approach for estimating the potential for unacceptable adverse environmental impact resulting from moving contaminated sediments to a relatively clean site by examining many factors other than just toxicity and chemical-specific concentrations. The effects on living resources which can be measured include an analysis of benthic alterations (i.e. changes in benthic community structure) and of more subtle changes to resident species (e.g. fish, crabs, molluscs) such as damage to external and internal tissues, and reproductive capacity. Rather than compare the results of contaminant sediment studies to proposed disposal sites, comparison would be made to reference sites to determine the magnitude of the hazard (see IMPLICATIONS 12 and 22).

TASK III. EVALUATE CHEMICAL-SPECIFIC RESULTS (EXPOSURE)

A. FINDINGS/OBSERVATIONS: The average concentrations of most metals in sediments to a depth of 150 cm are greatest at or near Site M, with decreasing levels upstream and toward the mainstem of the River.

B. FINDINGS/OBSERVATIONS: Two compound classes dominated the organic contaminants in the sediments of the Elizabeth River: the PNAs and the phthlates. Site O (in the vicinity of an abandoned creosote factory) exhibited far higher concentrations of PNAs and phthlates than other sites when the average sediment concentrations to a depth of 150 cm were examined. Average sediment concentrations of PNAs at Sites K to N were also elevated, particularly in the shoal areas.

IMPLICATIONS

17. The average concentrations indicate the potential magnitude of exposure to contaminants which would result if the sediments were disturbed during dredging to a depth of 150 cm or more. Depending on the extent of resuspension and spillage during dredging, violation of water quality standards could occur in the vicinity of the dredge site.

C. FINDINGS/OBSERVATIONS: Surface sediments (0-10 cm) in the Elizabeth River are still highly contaminated with certain heavy metals. Although some metals concentrations increased toward the more industrialized portion of the River, no single pattern consistently described the distribution of all metals in surface sediments.

D. FINDINGS/OBSERVATIONS: PNAs in sediments of the Elizabeth River continue to remain in high concentrations in surface sediments in certain regions of the Southern Branch: Sites K - Q.

IMPLICATIONS

18. The fact that surficial sediments are contaminated with certain potential toxicants indicates a route of exposure to living resources and the potential for impact to surface water quality. Therefore, patterns of surficial contamination should raise management concern. The potential for living resources and water quality to be affected by high concentrations of contaminants in sediments from 10-150 cm deep is minimal unless dredging activities or man-induced changes in hydrography would produce erosion patterns that would uncover the contaminated material. However, environmental assessments and management options based on concentrations alone may overlook toxic "hot spots".

E. FINDINGS/OBSERVATIONS: Five of the metals detected (Cd, Cr, Fe, Ni and Pb) appeared to decrease in the surface sediments when compared to previous studies. Copper and zinc showed no overall changes, while two regions (Site M, right shoal; and Site N, left shoal) displayed an increase in a number of metals.

IMPLICATIONS

19. In the absence of extensive "mapping" of surface metals concentrations, results from specific sites must be interpreted with caution. The observed temporal trends could be associated with more rigid enforcement and compliance with VPDES and VPA permits or changing use activities in the River.

F. FINDINGS/OBSERVATIONS: Unlike most of the metals, the organic contaminants did not display an overall decrease in surficial sediment. In certain regions, the surface sediment contamination increased from 1982 to 1989, while the concentrations in sediments from other areas tended to decrease. The most contaminated site (Site O) increased in heavy molecular weight PNAs but decreased in the lighter 2- and 3-ring organic compounds. Sediments from the urbanized lower reaches of the Southern Branch (Site K) increased in both low and high molecular weight PNAs.

IMPLICATIONS

20. The areas displaying an increase in surficial sediment contamination are quite likely in the vicinity of active sources of PNAs. The temporal pattern observed at Site K suggested active sources of combustion/petroleum contamination, possibly associated with shipping, marinas, shipyard activities, and/or urban runoff. On the other hand, the

temporal patterns at Site O suggested a source of "weathered" creosote contamination. However, sediment transport patterns appeared to be from the heavily industrialized region upstream towards this site, so the PNAs may not be attributed solely to the abandoned creosote factory in this region, but may be due to accumulation of particle-reactive contaminants from elsewhere as well.

TASK IV. RANK PRIORITY "HOT SPOTS"

A. FINDINGS/OBSERVATIONS: Although acute toxicity was not observed for sediments sampled in the Elizabeth River Long-term Monitoring (ERLTM) Phase I studies, triad plots did indicate impact along the other two axes (sediment chemistry and in situ effects) for certain sites: EBE1 (sediment chemistry and in situ effects); SBE1 and SBE2 (sediment chemistry); SBE3 (sediment chemistry and in situ effects); and WBE1 (metals and in situ effects).

B. FINDINGS/OBSERVATIONS: A simulation assessment of the triad approach indicated:

1. The calculation of RTR values produces highly non-normal distributions which are not amenable to the development of confidence limits by standard statistical techniques;

2. The calculation of RTR values for single samples may produce high probabilities of Type II error (i.e. the error of declaring a RTR value "non-impacted" when it is, indeed, from

an impacted site);

3. A bootstrap approach has been developed to produce confidence limits for triad plots for future studies.

IMPLICATIONS

21. Sites identified as most impacted by the triad approach should be considered among the "hot spots" identified by various components of the present study. However, since these triad plots were based upon relatively few samples (1-3 replicates), FINDINGS/ OBSERVATIONS IV.B.2 suggest that such a classification be treated as being preliminary in nature.

22. The triad approach represents a very important tool in the assessment of sediment quality since it incorporates not only sediment chemistry, but sediment toxicity (from laboratory studies) and <u>in situ</u> effects as well. It is recommended that triad plots incorporate confidence limits so that management actions can be based upon the degree of certainty that can be associated with each of the three components. Such a precaution would also necessitate that an adequate number of samples be taken to provide statistically defensible results.

C. FINDINGS/OBSERVATIONS: A comparison of 1989 data (ERLTM Phase I and Sediment Contaminant Distribution data sets) to AETs calculated for metals from a previous data base (USACOE study conducted 1979-1984) indicated that AETs for one or more metals were exceeded by sediments from the same areas that the triad plots indicated were most "impacted": EBE1, EBE2, SBE1, SBE2, SBE3 and WBE1. Other sites not evaluated by the triad, but observed to exceed the AETs were Sites EB, K, L, and M. All of these sites are located in the lower reaches of the Eastern and Southern Branches of the Elizabeth River.

D. FINDINGS/OBSERVATIONS: A simulation assessment of the AET approach indicated:

1. Correlations between toxic and "nontoxic" chemicals led to a very high probability of "false" AETs being established for the "nontoxic" chemical when the correlation coefficient (r) exceeded 0.5 (almost certainty above r=0.7);

2. Correlations above r=0.5 produced multiple "false alarms" when up to 300 samples were assessed; and

3. Correlations above r=0.6 produced AETs that had very little probability of being detected as false within a data base as large as 300 samples.

IMPLICATIONS

23. The sites identified as exceeding the 1979 AETs for metals should be considered among the "hot spots" identified by various components of the present study. However, the results of the simulation studies (FINDINGS/ OBSERVATIONS IV.D.) would suggest that the AETs be used only as "first cut" levels of

concern for identifying areas that display heavy contaminant loads of certain metals.

24. Sediment quality criteria based on the AET approach should not be implemented for regulatory use (e.g. as compliance or remediation targets) until they are validated for the particular ecosystem and contaminants of concern (e.g. through the use of spiked sediment bioassays; TIE investigations of pore water, etc).

E. FINDINGS/OBSERVATIONS: None of the sediments collected during the ERLTM Phase I study or the sediment contaminant distribution study exceeded the sediment quality "criteria" for PNAs calculated by the EP approach. In fact, the maximum concentrations of PNAs ever reported for the Elizabeth River would not have exceeded these "criteria", so the sensitivity of the approach would appear to be questionable.

IMPLICATIONS

25. The PNA concentrations which have been reported for sediments from certain areas of the Elizabeth River are among the highest ever reported in the world and a number of studies have demonstrated that these PNA-contaminated sediments produce a variety of acute and chronic toxic effects. Therefore, the EP approach would not appear to produce environmentally protective criteria if they are not exceeded for these highly

impacted conditions.

26. Considering that neither the AET (FINDINGS/ OBSERVATIONS IV.E.) nor the EP (FINDINGS/OBSERVATIONS IV.D.) approaches produce completely defensible criteria for contaminants in sediments, an alternate approach to sediment quality assessments should be based upon toxicity rather than numerical criteria. Sediment demonstrated to be toxic could be further investigated in a TRE/TIE scenario to identify contaminants of concern and their sources for the purposes of control/remediation.

27. Such an approach could be implemented as part of the VPDES and VPA permit program:

a. Industries with discharges (point sources) or land use activities (potential nonpoint sources) that are suspected of contributing toxics to an estuary would be required to monitor the sediments in the vicinity of their operation(s) for toxicity and/or all three triad components.

b. This information would contribute to a system-wide data base that would allow the assessment of whether the sediments contain "toxics in toxic amounts", and would provide managers the ability to perform trend analyses on the entire system.

c. If sediments are shown to be toxic, the "burden of proof" would shift to the industry to conduct special studies (TRE/TIE scenario) that would establish the contaminants and/or source activities responsible for the toxics in the system.

d. If the industry's activities are responsible for the toxicity (i.e. the contamination cannot be demonstrated to have been transported from elsewhere), tighter controls, and/or BMPs would be implemented.

e. Routine (e.g. annual) monitoring of the surface sediments for toxicity and the contaminants of concern would be required for demonstrating compliance.

f. If sediments are shown to produce severe impacts on living resources <u>in situ</u> and control strategies do not produce an adequate decrease in toxicity, remediation may be necessary to reduce toxic impacts in the system.

TASK V. DETERMINE RISK TO LIVING RESOURCES AND WATER QUALITY

A. FINDINGS/OBSERVATIONS: Flux rates were determined for 6 PNAs in sediments from the Elizabeth River. The flux rates were moderately low, ranging on the average from 86 $\mu g/m^2/day$ to 611 $\mu g/m^2/day$.

IMPLICATIONS

28. The measured flux rates and resulting empirical models suggested that sediment fluxes of PNAs into the water would not alone lead to violation of water quality criteria (naphthalene, acenaphthene, fluoranthene). Rather than basing management decisions on numerical criteria alone, the focus should be on the organic "toxics in toxic amounts" that accumulate in sediments when control strategies fail.

B. FINDINGS/OBSERVATIONS: The flux rates of the nonpolar organic contaminants were found to depend upon the log P of the contaminants and the organic carbon normalized concentrations in the sediments. The empirical data fit theoretical relationships quite well and predictive numerical models were developed.

IMPLICATIONS

29. Once the flux equations are validated, they can be used to estimate the magnitude of input of nonpolar organic contaminants from sediments as a nonpoint source for incorporation in loading inventories established as integral components of system-wide control strategies.

C. FINDINGS/OBSERVATIONS: Six metals (Cd, Cu, Fe, Mn, Ni and Zn) had detectable sediment fluxes under the environmental conditions observed in the Elizabeth River during late spring. Although flux rates were variable, the average rates for all metals at all sites tended to be positive, indicating that the sediments were acting as a source of metals to the water column. The average daily flux of metals generally represented 1-12% of the ambient load in the water column, to produce doubling times of 8-86 days. The only major exception to the pattern was observed for sediments from Site EB which displayed a negative flux (influx into sediment) of copper.

IMPLICATIONS

30. Considering the rather long residence time of water in the River, the modest positive flux rates of metals may be significant sources to the overall loading of the water column. Therefore, they should be taken into account when loading inventories are calculated. Since metal flux dynamics are highly dependent on a variety of environmental conditions, rates should be determined at other sites and for other seasons before comprehensive loading inventories can be estimated.

31. The negative flux of copper at Site EB may be indicative of a recent source of copper in the water which has yet to reach equilibrium with the sediments. The temporal comparison of data collected in 1974 and 1989 in this region also tends to indicate an increase of copper in surface sediments, suggesting less than effective control strategies for this metal and/or a recent change in use patterns (e.g. a change from lead-based or TBT-based to copper-based antifouling paints) in the region.

Conversely, the positive fluxes for the other metals may be seen to support the temporal pattern assessment that indicated that many of the metals in the surficial sediments tended to be decreasing over time. Thus, more effective control strategies and/or changing use patterns (e.g. use of unleaded fuel; banning of lead in paints, etc.) may be reflected in less metals in water, higher sediment flux rates, and, ultimately, lower concentration in the surficial sediments (due to leaching and/or burial). However, this interpretation must be tempered by the previously stated caveats concerning the preliminary nature of the flux measurements and the potentially significant effects of seasonal changes in environmental conditions or flux dynamics.

D. FINDINGS/OBSERVATIONS: Ammonium fluxes were always positive and generally quite high. The average daily percent contribution to ambient water column loads ranged from 2-6%. Ammonium fluxes during late summer, when water temperatures are higher and bottom waters become hypoxic or anoxic, would be expected to be even higher.

IMPLICATIONS

32. As with the metals, the magnitude of ammonium flux rates must be considered in the context of loadings from other sources in the system. If maximum sediment fluxes expected during the late summer months (particularly in the deep

channels) tend to dominate the input by other loading, sources control strategies may have to shift from direct point source control of ammonium to a more general control strategy for reducing the input of organic matter (BOD) to the system, and/or to remediation of the sediments displaying the greatest flux rates.

33. Flux rate measurements and numerical models can also be used to estimate the magnitude of leaching processes in diminishing surficial sediment contamination. Such indices as halving times would be useful in risk assessments (exposure assessments) and remediation policies.

TASK VI. EVALUATE RECENT CONTAMINANT LOADINGS

IMPLICATIONS

34. The results from this report cannot provide comprehensive estimates of the magnitude of recent contaminant loadings, indications of sources, and/or definitions of trends for all sites in the Elizabeth River. The resource manager must draw upon the available scientific and regulatory data to determine whether significant contaminant loadings are continuing.

Although there are some indications from temporal and spatial (depth) patterns that loadings of certain of the contaminants have changed (see FINDINGS/OBSERVATIONS III.C-F.) over the past decade, the responsible mechanisms are not clear. The observations that surface sediments continue to remain highly contaminated with certain organic and inorganic chemicals does suggest that sources are still active, although the "memory" of these surficial sediments may be quite long in certain areas (see FINDINGS/ OBSERVATIONS VIII A-B and IMPLICATION 43).

TASK VII. DEVELOP MORE STRINGENT POINT AND NON-POINT SOURCE POLLUTANT CONTROL STRATEGIES

A. If the data provides evidence of recent contaminant loadings to the sediment (see IMPLICATION 34), more effective pollution control strategies would be required. The following are options that may be considered:

IMPLICATIONS

35. Contrary to the Draft Lower James River Basin Water Quality Management Plan (VWCB, 1990b), some Elizabeth River segments or sites may require wasteload allocations (WLAs), load allocations (LAs) and total maximum daily loading (TMDL) models due to the magnitude and nature of suspected and proven toxicants found in this Water Quality Limited area. The development and enforcement of point and non-point source controls using limits based on TMDLs would aid in reducing future contaminant loadings to the sediments.

36. In the development of TMDLs, the contribution of pollutants resulting from construction and dredging activities conducted in the Elizabeth River system should receive special attention.

37. Contaminant loadings to the sediments, particularly in segments identified by Task IV as posing an unacceptable risk should be considered in the development of Stormwater Management Plans, BMPs and other non-point source control measures.

38. Site-specific management approaches for reducing or eliminating the exposure from in-place contaminants must be based on adequate knowledge of the source <u>and</u> the mechanisms (contaminant transport through the water column and through sediment transport) which may deposit or partition these materials into the sediments.

B. In areas where the risk posed by contaminated sediments is considered unacceptable, the contribution of sediment contaminant loadings could be delegated to the owner of the source as a monitoring program requirement which is designed for compliance with CWA goals and for long-term trend assessments (see IMPLICATION 27).

39. Requirements could be added to VPDES and VPA permits for annual monitoring of toxicity or bioaccumulative potential in

sediments from mixing zones or zones of initial dilution.

40. Special studies could be mandated when unacceptable risks are posed by the sediments in mixing zones to enable source identification, development of remediation plans and addition of more stringent discharge controls.

41. An extensive survey should be conducted to determine if all municipal and industrial activities are monitored under the VPDES and VPA permit program, and to ensure that all non-point source land use activities have been assigned VPA permits which include BMPs.

TASK VIII. EVALUATE RISK REDUCTION BY NATURAL PROCESSES

A. FINDINGS/OBSERVATIONS: A preliminary geochronology study of four cores from the banks of the Elizabeth River indicated that these shallow areas have low sedimentation rates (0.8 to 2.4 mm/year) which appeared to increase in an upstream direction. Therefore, the shallow areas along the most highly industrialized areas of the mainstem and Southern Branch do not appear to be a major sink for particle reactive pollutants. However, as the navigational channel was approached (in what has been referred to as "shoal" areas), the depositional rates appeared to be at least an order of magnitude higher (>1 cm/year), suggesting that the channel may be a region of preferential accumulation of particle reactive pollutants. A previous investigation by Nichols and Howard-Strobel (1986) suggested that landfill activities over the last 150 years have reduced the tidal prism and current velocities to the point that sedimentation rates in routinely dredged channels may be one to two orders of magnitude still higher (15-195 cm/year).

B. FINDINGS/OBSERVATIONS: Results of grain size analysis indicated that the grain size decreases upstream, tending to support the patterns observed in the geochronology study: fine grain sediments appeared to be transported upstream, particularly along the channel. Although the fate of particle reactive pollutants that are transported by non-tidal circulation (i.e. movement upstream along the salt wedge) is unknown, it is suspected that a major "sink" (i.e. the probable site of deposition) would be associated with the yet to be delineated turbidity maximum zone.

IMPLICATIONS

The low sedimentation rates observed in the shallow areas along the banks of the River in the regions studies may have several implications:

42. Contaminants in sediments found in the shallow areas may reflect input from local sources (point sources or land use activities) while those in the channels, particularly as one moves upstream, may have been transported from elsewhere (more distant sources). Therefore, hydrography and sediment

transport processes have to be taken into account if sediment quality issues are to be incorporated into the regulatory process (i.e. source identification, control and compliance; see IMPLICATIONS 27, 35-41).

43. Sediment contaminants in shallow areas may not be readily removed from the system (i.e. from surficial layers subject to chemical exchange with the water column and/or biological exposure) by sedimentation (burial) processes. If the rates measured are representative, it may take 10-20 years to remove the contaminants from the sediment-water interface to a depth of 2 cm below the surface. Of course, processes such as weathering (i.e. breakdown/detoxification of contaminants by biogeochemical processes) and leaching (i.e. flux processes) could also greatly influence the capacity of the sediment from those regions to be "cleaned up" after sources are eliminated. Management decisions concerning remediation would have to assess these processes as well.

44. Particle-reactive pollutants transported into channels may be removed by dredging if: sources are controlled, "sink" areas are identified and targeted; the dredging operations are shown to be a cost-effective response to the hazard in the risk assessment process (if not already mandated for navigational purposes); and the dredging is shown not to produce a slumping of contaminated deeper sediments from shoal areas back into the channels. It is possible that maintenance dredging and/or

channel deepening operations could remove much of the contaminants from the sediment of the system if the dredging operations do not reintroduce the fine silt-clay particles (most likely to be associated with particle-reactive pollutants) back into the water column. However, it should be noted that the River is routinely dredged and, while the contamination/toxicity of the sediments <u>is</u> initially reduced, it appears to return, so sources apparently have not been completely controlled.

TASK IX. EVALUATE FOR DISPOSAL IN ACCORDANCE WITH EXISTING REGULATIONS

<u>IMPLICATIONS</u>

45. The decision to remove and dispose of in-place contaminants must be based on an evaluation of a number of issues:

a. The resource manager must consider the alternatives to dredging of contaminated materials and evaluate disposal site options, if dredging is warranted.

b. The evaluation of dredging, transport and disposal techniques assumes that sufficient information is available to perform an environmental risk assessment for each option. The tiered assessment guidelines presented by the EPA/ACOE (1976, 1977 and draft 1990) provides an approach which is used in the evaluation of dredged materials for disposal in navigable and ocean waters. A priority ranking of the sediments based on whether they present no potential impact or a potential for unacceptable adverse impact is provided in these guidelines and at 40 CFR 230.

c. Prior to completing the decision process, the manager must understand mechanisms for transport of trace contaminants into sediments and mechanisms of release, as well as the influence on water quality, impact to living resources (toxicity and bioaccumulation), and the potential risk to human health. This information will aid in the assessment of risk posed by sediment left in place, or by sediment removal, transport and disposal. Additionally, the long-term effects at the disposal site must be determined and monitored.

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